8. UNCERTAINTY

8.1. INTRODUCTION

This chapter addresses uncertainty in dioxin exposure assessment performed with the methodologies presented in this document. Some discussion of the issues commonly lumped into the term "uncertainty" is needed at the outset. The following questions capture the range of issues typically involved in uncertainty evaluations:

(1) How certain are site specific exposure predictions that can be made with the methods?
(2) How variable are the levels of exposure among different members of an exposed local population?
(3) How variable are exposures associated with different sources of contamination?

The emphasis in this document is in providing the technical tools needed to perform site-specific exposure assessments. For the assessor focusing on a particular site, question (1) will be of preeminent importance. Therefore, the emphasis of this Chapter is to elucidate those uncertainties inherent to the exposure assessment tools presented in this document. This chapter examines the uncertainties associated with estimating exposure media concentrations of the dioxin-like compounds using the fate, transport, and transfer algorithms, and also identifies and discusses uncertain parameters associated with human exposure patterns (contact rates and fractions, exposure durations, etc.).

Section 8.2 focuses on uncertainty issues associated with the use of the ISCST3 model for air transport modeling for the stack emission source category. The ISCST3 model and its application in this assessment are presented in detail in Chapter 3. Section 8.3 discusses the variability and uncertainty with chemical-specific parameters which are required for all source categories of this assessment methodology. Section 8.4 provides a general overview of all key uncertainties with each pathway.

A site specific assessment will also need to address the variability of risks among different members of the exposed population, the second key question above. The level of detail with which this can be done depends on the assessors knowledge about the actual or likely activities of these residents. In this document, one approach to evaluating this variability is demonstrated. Separate "central" and "high end" scenario calculations are presented to reflect different patterns of human activities within an exposed population. "Central" scenarios are constructed to represent typical behavior patterns for residential exposures in a hypothetical rural

setting. "High end" calculations focus on a farming scenario where individuals raise food for their own consumption, in the same rural area. It should be emphasized that high end calculations could also have been developed for residential exposures by making, for example, higher range assumptions about the duration of residence or contact rates with the contaminated media. Indeed, this would be recommended for an assessment where considerable emphasis was placed on residential exposures. The key issue with regard to intra-population variability is that it is best (if not only) addressed within the context of a specifically identified population. If such information is available, a powerful tool that can be used to evaluate the variability within a population is Monte Carlo Analysis. Section 8.5. reviews recent Monte Carlo studies which have been done for exposure to 2,3,7,8-TCDD. Assumptions on distributions of exposure patterns and fate and transport parameter distributions are described, as are the results of their analyses. Aside from this review, this chapter does not address question (2) in any further manner.

With regard to question (3), this document does not present a detailed evaluation of how exposure levels will vary between different sources of release of dioxin-like compounds into the environment. Volume II of this assessment examines sources of release of dioxin-like compounds into the environment. This document, Volume IV, presents methodologies for three types of sources - soil, stack emissions, and effluent discharges into surface water bodies. While this document demonstrates the methodologies developed for these sources with source strengths and environments crafted to be plausible and meaningful, there is still a great deal of variability on both the source strengths and on the environments into which the releases occur. For example, the frequency with which farms and rural residences are near stack emissions of dioxin-like compounds is not addressed. The scenario calculations in Chapter 5 are intended to be illustrative; the exposure levels that are obtained there are not intended to be typical of actual exposures for the sources and pathways assessed.

Nonetheless, some readers might ideally wish information on both the magnitude of actual exposures and the variability of these exposures associated with different sources of dioxin-like compound releases into the environment. However, the analysis presented in this chapter cannot support so broad a goal. Representative data to address the variation of dioxin exposures are becoming available for sources as well as exposure media. Volume II discusses and quantifies releases from known sources in the US, and the compilation of environmental and exposure media concentrations presented in Chapter 3 of Volume III of this assessment displays the range of measured concentrations in the environment. The careful selection of certain literature reports on concentrations of dioxin-like compounds to represent background conditions, described in Chapter 4 of Volume III, is one way such environmental measurements

can be used. References to EPA and other assessments on dioxin-like compounds have been made throughout this document, such as those related to soil exposures (Paustenbach, et al., 1992a), exposures to contaminated fish (EPA, 1991a), exposures resulting from land disposal of sludges from pulp and paper mills (EPA, 1990), just to name a few. Still, studies comparing and ranking different sources and exposure patterns, and elaborations on ranges of source strengths and exposures, are generally not available. Information in Volumes II and III of this assessment, and procedures for source specific evaluations in Volume IV, can provide others with information and tools to begin such analysis.

8.2. A DISCUSSION OF UNCERTAINTY ISSUES ASSOCIATED WITH THE USE OF ISCST3 FOR TRANSPORT AND DISPERSION OF STACK EMITTED CONTAMINANTS

Air dispersion and deposition analysis was performed using the ISCST3 Model. The model is intended to provide long term average air concentrations and wet and dry deposition flux. This section discusses some of the uncertainties and critical parameters associated with the general modeling approach used in ISCST3, and reviews some of the literature on model testing and validation.

Atmospheric dispersion in ISCST3 is modeled using the common Gaussian plume model. Downwind concentrations of the dioxin-like chemicals are calculated as a function of stack height, the mass emission rate, the wind speed, and general atmospheric conditions. The Gaussian model assumes that the emission concentrations predicted by the model will fit a normal distribution. The principal assumptions in the Gaussian model are (Kapahi, 1991):

- The air concentration of the chemical at a fixed distance from the source is directly proportional to the emission rate from the source;
- The air concentration of a given chemical is inversely proportional to the wind speed corresponding to the effective height of release of the chemical into the air;
- The predicted ground-level concentration of the chemical approaches zero at large distances from the initial point of release.
- The model is steady-state.
- The model assumes constant wind speed, wind direction, and atmospheric stability over time and space for a given time period.

In general, the Gaussian plume model has been shown to predict annual average ambient air concentrations of a chemical emission from an industrial source to within a factor of one-order of magnitude of measured values, and in some cases, within a factor of 3 to 4-fold of field

measurements (Cohrssen and Covello, 1989). This modeling error spans both sides of the predicted concentration, that is, the actual concentration may be plus or minus this amount of the predicted value. Even more assertive, an early position paper on the application of gaussian short-term dispersion models claimed an approximate factor-of-two accuracy in the absence of complicating factors (complex terrain, building wake effects) (AMS, 1978).

The most sensitive aspects to variability in modeled predictions of ambient air impacts, if emissions are held constant, are stack height (height of the release), and terrain (flat verses complex topography). To investigate modeling variability, EPA placed a prototype hypothetical hazardous waste incinerator in flat terrain and elevated terrain in geographical areas around the U.S. (EPA, 1991b; analysis conducted with the Industrial Source Complex, or ISC, model). Then the stack height was varied at these particular locations. Numerous runs were made at twelve specific sites to compare and contrast the influence of stack height and terrain on predicted ambient air concentrations of various mass emission rates of specific inorganic pollutants. A series of tables were developed from this sensitivity analysis from which the numerical estimation of the variability as a function of stack height and terrain can be inferred. When the hypothetical hazardous waste incinerator was modeled in flat terrain, e.g., topography within a distance of 5 km is not above the height of the stack, and the stack height was varied from 4 meters to 120 meters, the variability in the predicted ambient air concentration spanned two orders of magnitude (100). The lower stack height resulted in a predicted ambient air concentration that was 100 times greater than the concentration predicted using the tallest stack height. When the hypothetical hazardous waste incinerator was located in complex terrain over the same range of physical stack heights, the variability in estimated groundlevel concentration of the subject pollutant spanned two orders of magnitude (100-fold). In the latter case the stack height was computed as the terrain-adjusted stack height by subtracting from the physical stack height the influence of terrain on plume rise. From the limited sensitivity analysis of hazardous waste incinerators, it can be assumed that the predictions of spacial ground-level ambient air concentrations of dioxin-like compounds could differ from values in Tables 3-17 and 3-18 by two-orders of magnitude in consideration of changes in stack height or changes in terrain. For example, Tables 3-17 and 3-18 show that the maximum annual average ambient air concentration of 2,3,7,8-TCDD predicted near the hypothetical incinerator is approximately 10^{-11} $\mu g/m^3$ for the stack height of 30.5 meters, and assuming flat terrain. If only the stack height is varied from 20 meters to 120 meters, and all other modeling parameters are held constant, then the predicted ambient air concentration would be approximately 10 times greater and 10 times less than the estimated concentration, respectively. The uncertainty is broader when considering the influence

of topography on predictability of the ground-level concentrations from the model. If only terrain elevation is varied at a distance of 5 km from the hypothetical incinerator from zero elevation to 30.5 meters, e.g., the height of the stack, then the predicted ambient air concentration of 2,3,7,8-TCDD would be approximately ten times greater. The tables derived in the hazardous waste incineration analysis have a limitation of elevation of terrain to the height of the stack.

The most uncertain aspect to the modeling is the estimation of dry and wet deposition flux of dioxin-like compounds on the vicinity of a hypothetical incinerator. Contributing most to this uncertainty seems to be the settling velocities and scavenging coefficients estimated for specific particle size diameters (Cohrssen and Covello, 1989; Doran and Horst, 1985). Seinfeld (1986) found that particles over 20 microns in diameter settle primarily by gravity, whereas smaller particles deposit primarily by atmospheric turbulence and molecular diffusion. Considerable, but non-quantifiable, uncertainty exists with respect to deposition velocities of particles 0.1 to 1.0 microns in diameter (Seinfeld, 1986). The uncertainty is difficult to define. The wide variation of predicted deposition velocities as a function of particle size, atmospheric turbulence and terrain adds to this uncertainty (Sehmel, 1980). However, Gaussian plume dispersion models have been field validated for their ability to spatially predict dry deposition flux over some specified distance (Doran and Horst, 1985). In a series of field experiments conducted by Pacific Northwest Laboratory (Doran and Horst, 1985), zinc sulfide was used as a depositing tracer gas, and sulfur hexafluoride was used as a non-depositing tracer gas to compare and contrast modeling results with field measurements of dry deposition and atmospheric diffusion of the gases. The tracer was released from a height of 2 meters, and all releases were made under relatively stable atmospheric conditions. Five sampling stations were located downwind of the release from 100 to 3200 meters. The results of these experiments showed good agreement with the predicted verses the measured deposition of the tracer ZnS. The overall correlation coefficient between predicted and measured deposition concentration was found to be 0.82 (Doran and Horst, 1985), but the models marginally over-predicted deposition flux near the source of release, and under-predicted deposition flux at 3200 meters.

Travis and Yambert (1991) have evaluated the uncertainty in modeling the dry deposition flux of particulates using four standard Gaussian plume dispersion models. Since deposition flux is dependent on deposition velocity for a given particle mass and diameter, comparisons were made between model-generated deposition velocities and measured values found in the open literature for particles ranging from 0.01 to 30 microns in diameter. It was found that measured deposition velocities for a given particle size in the scientific literature exhibit variability spanning roughly two orders of magnitude. The analysis of the mean predicted deposition

velocities to mean measured values showed that most measured data exceeded the predicted data for all four models. Moreover, the models underestimated the mean deposition velocities for particles in the range of diameters from 0.05 to 1.0 microns.

Similar uncertainty probably exists with regard to scavenging of various diameter particles by various intensity of rainfall. Seinfeld (1986) has calculated scavenging coefficients in terms of the removal efficiency of particles of a given size by rain droplets having a given momentum. Seinfeld (1986) found that the scavenging coefficient of a given particle diameter corresponding to a given rainfall intensity can be calculated based on physical laws, but there is a complete absence of research data to verify these calculations. Hence it is not possible to address the accuracy nor uncertainty of the wet deposition flux estimated in Table 3-19.

There have been some limited validation work done with ISCST3 and its ISC predecessors. Chapter 7 described a model validation exercise for air dispersion and deposition/soil concentration modeling done for dioxins in the vicinity of a municipal solid waste incinerator known to be emitting large amounts of dioxins. The predicted concentrations were mostly within a factor of 10 of observations, higher or lower, for both air and soil. There was evidence that the profile of dioxins in both the air and the soil were distinct from the profile of dioxins being emitted from the incinerator. This observation suggests transformations in the dioxin profile in either, or both, the air and soil environments. In clearly impacted ambient air samples that were downwind of the incinerator during sampling events, for example, the measured profile suggested a more predominance of lower chlorinated dioxins than was seen in the stack emission. Two explanations were offered to explain this observation: the higher chlorinated dioxins deposited much more so than the lower chlorinated dioxins, which lessened their predominance in the profile and/or higher chlorinated dioxins dechlorinated to form lower chlorinated dioxins. When testing air dispersion alone (no deposition, no atmospheric decay or transformation of emitted dioxins), the air concentration profile perfectly matched the stack emission profile, as it should, so neither of these possibilities could be tested. However, when testing the deposition/soil concentration capabilities of ISCST3, evidence did strongly suggest that the model was underpredicting the deposition rate of OCDD, at least. Even with this possible finding, the disparity between the soil concentration profile and the stack emission profile continued to suggest that transformations may be taking place in soils and/or the air which were not captured in the model testing at this site. In general, the model was able to duplicate the trend of elevations in both air and soil near the facility, to within a factor of 10 of these elevations.

Early ISC (the predecessor to ISCST3) model validation work was conducted by Bowers, et al. (1981). They tested the gravitational dry particle deposition algorithms, new at that time, and showed that the model predicted deposition rates generally within a factor of two of measured depositions of glass microspheres of 50 to 200 µm measured in an experimental setting. They also tested the capabilities of building wake effects using data from diffusion experiments conducted at a Nuclear Power Station in which the tracer SF_6 was released from the reactor building main vent and the tracer Freon 12B2 was simultaneously released from three vents on the adjacent turbine building. They then predicted concentrations of these tracers with and without building wake effects, and found that the inclusion of building wake effects improved the average correspondence between modeled and observed concentrations by almost a factor of 2.

8.3. UNCERTAINTIES AND VARIABILITIES WITH CHEMICAL-SPECIFIC MODEL PARAMETERS AND ASSUMPTIONS

This assessment assumed that levels of dioxin-like compounds in soil and sediment were constant over the period of exposure, with two exceptions. One circumstance was when contaminated soil eroded from one site and deposited on a site of exposure nearby - the soil contamination source category. The other was when stack emitted particulates deposited onto a site of exposure - the stack emission source category. In both these instances, it is assumed that only a relatively thin layer of surface soil at the site of exposure would be impacted, and that this thin layer is subject to dissipation processes - erosion, volatilization, possibly degradation. Data in Young (1983) implied a soil half-life of 10 years for surficial 2,3,7,8-TCDD residues, although the circumstances of the soil contamination were not analogous. Specifically, a 37 ha test area at the site had received an estimated 2.6 kg of 2,3,7,8-TCDD over a two year period. Soil sampling which occurred over 9 years from the last application suggested that less than 1 percent remained at the test area. Although Young hypothesized that photodegradation at the time of application was principally responsible for the dissipation of residues, other mechanisms of dissipation including volatilization, erosion, and biological removal may also have contributed to the loss of residues. Soil sampling over time after application implied a dissipation half-life of 10 years for soil residues of 2,3,7,8-TCDD. Mclachlan, et al. (1996) reported on an analysis of soil taken from experimental plots which had been amended with sewage sludge in 1968 and sampled in 1972, 76, 81, 85, and 90. These archived samples were analyzed for all 17 dioxin-like CDD/Fs, and based on an analysis of results, McLachlan and coworkers concluded that half-lives were on the order of 20 years, with dioxin removal from the plots being mainly physical removal

processes (overland runoff, wind erosion). Furthermore, their results suggested that all congeners had been removed at roughly the same rate, which is why they concluded that removal processes were mainly physical and very little in-situ degradation appeared to be occurring. Paustenbach, et al. (1992a) reviewed several reports of the soil dissipation of 2,3,7,8-TCDD, including Young (1983), and concluded that the half-life of 2,3,7,8-TCDD residues below the surface varied from 25-100 years. A half-life of 25 years ($k = 0.0277\ yr^{-1}$) was assumed to apply to all dioxin-like compounds in this assessment.

Section 2.6.1, Chapter 2 in Volume II of this assessment, reviewed the literature on degradation of dioxin-like compounds. As discussed, biological transformations as well as chemical processes (oxidation, hydrolysis, and reduction) do not appear to result in substantial degradation of these compounds. There is evidence of photolysis, particularly when dissolved in solution and when organic solvents are present. Most of these data are specific to 2,3,7,8-TCDD. Uncertainty is introduced into parameter assignment when information specific to one congener is assumed to apply to all dioxin-like congeners. However, it is judged that there is no good data available to assign different soil dissipation rates to different dioxin congeners in this assessment, and McLachlan's (1996) data is judged to be reasonably strong to support an assumption that all dioxin congeners dissipate with roughly the same half-life.

Dissipation of surficial residues could translate to lower soil-related exposures including particulate inhalations, soil ingestion, and soil dermal contact. However, it is not clear that reductions in exposure would, in fact, occur, particularly if the soil is contaminated below the surface. Processes such as wind erosion, soil erosion, or volatilization originating from deeper in the soil profile, could serve, in a sense, to replenish reservoirs at the soil surface. Depositions back onto soils from other soils, or depositions from distant sources, also replenish soils. Given very low rates of degradation (for all degradation processes except photolysis), the assumption of no degradation for the soil contamination source category is reasonable with moderate, but unquantifiable uncertainty.

In evaluating an assumption of no degradation, another issue to consider is the depletion of the original source of contamination. For the stack emission and effluent discharge source categories, the assumption is made that steady releases occur while the source is active. Therefore, depletion of the original source is not an issue. For the soil contamination source category, it is assumed that the reservoir of contaminant is constant throughout the duration of exposure. If such a duration is assumed to be very long, then degradation or dissipation of soil residues would be more critical than if the duration were relatively short. Uncertainties associated with the duration of exposure are discussed in Section 8.4 below. Also, Section 6.4 in

Chapter 6 evaluated the assumption of a constant soil concentration by estimating the time it would take for a 15-cm reservoir of soil contamination to be depleted, using the dissipation algorithms of this assessment. These algorithms include volatilization, soil erosion, and wind erosion, with lesser releases due to biological uptake, and leaching and runoff. It was found that it would take over 90 years to deplete a 15-cm reservoir, lending some credibility to a non-degradation assumption if the exposure duration were in the range assumed for the demonstration scenarios of this assessment, 30 years.

A critical contaminant parameter required for the procedures in this assessment is the octanol water partition coefficient, Kow, although none of the fate and transport algorithms directly require a Kow. One of the empirical biota transfer parameters is, however, a function of Kow. This is the RCF, or Root Concentration Factor, which estimates the transfer of contaminant from soil water to root. Log Kow estimates for dioxin-like compounds range from 6.00 to 8.5, with higher log Kow associated with higher chlorination. However, this is not a certain parameter. Estimates in literature for 2,3,7,8-TCDD, for example, range from 6.15 to 8.5. The uncertainty of the RCF is addressed in Chapter 7, Section 7.3.9, where experimental data on the transfer of dioxins from soil to carrots was used in a validation exercise. It was found that the RCF allowed for the reasonably accurate simulation of the transfer of dioxins to the carrot peel, with the model able to predict peel concentrations within a factor of 2 for 15 of 20 observations, and for the other five observations, predictions and observations differed by a factor of 5 or less.

Two biota transfer coefficients are used to estimate fish tissue concentrations based on water body sediment concentrations: the Biota Sediment Accumulation Factor, BSAF, and the Biota Suspended Solids Accumulation Factor, BSSAF. There are no empirical relationships which estimate these as a function of the more common Kow for dioxin-like compounds. Rather, values were assigned based only on experimental and field data. Needless to say, most of the data available was for 2,3,7,8-TCDD, leaving large gaps for other compounds. Also, there is no data available for estimating the BSSAF, a parameter proposed in EPA (1993) which was used in the effluent discharge source category. The BSSAF was set equal to the BSAF for this assessment. Field data including bottom sediment concentrations and concurrent fish concentrations were used to determine values for BSAF. The limited field data available for BSAF suggests values in the range of 0.03 to 0.30 for 2,3,7,8-TCDD, with higher values approaching 1.00 indicated for bottom feeders (catfish, carp, etc.), and decreasing values as the degree of chlorination increases - limited information suggests values in the 10^{-3} to 10^{-2} range for hexa- through octa- CDDs and CDFs. EPA (1995) used available data to develop the “bioequivalency factors”, BEFs, or multipliers to the BSAF or BSSAF to assign values for

congeners other than 2,3,7,8-TCDD, when data on only 2,3,7,8-TCDD is available. The BEF concept and the BEFs are described further in Chapter 4. They were used to assign values for the BSAF/BSSAF for other dioxin-like congeners assuming a BSAF/BSSAF of 0.09 for 2,3,7,8-TCDD. Data on PCBs suggest that BSAFs are higher than those of CDDs and CDFs by an order of magnitude and more, and that the trend with increasing degrees of chlorination is not the same. The data indicates that BSAFs for PCBs increase from dichloro- through hexa- or perhaps hepta-chloro PCBs, and decrease thereafter.

A bioconcentration factor, BCF, translates the average contaminant in the diet of the cattle into a beef or milk fat concentration. Experimental rather than field data was available for estimates of BCF for dioxin-like compounds. Farm animals were fed known quantities of these compounds and their body tissues and milk were monitored over time to arrive at BCFs. Data showed that the BCF decreased to below 1.0 as the degree of chlorination increased. An experimental data set, including analysis of 16 of the 17 dioxin-like congeners, described in McLachlan, et al (1990), was used to assign BCF values for this assessment. A more recent study, by Fries, et al. (1999), developed BCFs for 14 of 17 congeners in a feeding experiment where four cows were fed PCP-contaminated wood. Results showed a good agreement between these BCFs and those developed from the data of McLachlan, et al. (1990), although the BCF for 2,3,7,8-TCDD was highest in this experiment at 7.1 as compared to the BCF of 5.76 developed from McLachlan's data and used in this assessment. Limited data showed PCB BCFs to be the same order of magnitude, although trend data for increasing degrees of chlorination was not available.

Similar bioconcentration factors, also termed BCF in Chapter 4, were described for chicken fat. Like the beef/milk fat BCF, they were developed from experimental data on chickens and eggs (Stephens, et al., 1995). The transfer of vapor-phase dioxins from air to plant is also modeled with a simple biotransfer factor, termed B_{vpa}, is also developed from field data.

Obviously, a degree of uncertainty is introduced when relying on these empirical bioconcentration or biotransfer coefficients to estimate concentrations in fish, beef, milk, chicken, eggs, and terrestrial vegetation. The variability in the data suggests up to an order of magnitude range of variation may result from use of these parameters. All but one of these factors (the RCF) were developed from field or experimental data on dioxin-like congeners or homolog groups. This, by definition, will lend a degree of credibility to their assignment. Also, a validation exercise described in Chapter 7 testing the air-to-beef algorithm is a test of two of these biotransfer/bioconcentration factors, the B_{vpa} and the BCF, and both appear to be supported by this exercise. It appears likely, therefore, that the actual variation in these

biotransfer/bioconcentration factors, is less than an order of magnitude, perhaps less than a factor of five.

Another important chemical-specific parameter that can be estimated from Kow or estimated experimentally is the organic carbon partition coefficient, Koc. Koc describes the steady state partitioning between soil or sediment organic carbon and water; it impacts the volatilization flux from soils, and the partitioning between suspended sediment and water in the water column. Koc is used to estimate in-situ partitioning using a fraction organic carbon in the soil or sediment, OC_{sl}, OC_{sed}, and OC_{ssed}, as $Koc*OC_{sl}$, etc. The resulting chemical-specific parameter is termed the soil (or sediment) partition coefficient, Kd_s (or Kd_{sed}, Kd_{ssed}). The empirical equation used to estimate Koc from Kow in this assessment was derived by Karickhoff (1979). This equation was chosen over others available (Lyman, 1982) because it was derived from laboratory testing of 10 hydrophobic contaminants. Others available would have led to lower estimates of Koc. The Koc for 2,3,7,8-TCDD estimated for this assessment using Karickhoff's relationship was 3,980,000. Some data implies that this estimate itself may be low for 2,3,7,8-TCDD. Studies reviewed in Section 2.4.5., Chapter 2 of Volume II of this assessment, particularly those Jackson, et al. (1986) and Lodge (1989), indicate 2,3,7,8-TCDD Koc estimates in the range of 20,000,000 to greater than 30,000,000.

Another contaminant parameter is the Henry's Constant. Volume III, Chapter 2, provides the values of the Henry's Constants, H, for dioxin-like compounds, some of which were estimated given vapor pressure and water solubility data. The CDD/F Henry's Constants were in the 10^{-6} to 10^{-5} atm-m^3/mol range, while coplanar PCBs were in the 10^{-5} to 10^{-4} range, with one high value at $3x10^{-3}$ atm-m^3/mol.

Finally, the contaminant molecular diffusivity in air is required for estimates of volatilization flux from soils. The molecular diffusivity in air is set at 0.05 cm^2/sec for all dioxin-like compounds. Molecular diffusivity is a property of both the chemical and the medium. It represents the propensity of a chemical to move through a medium. It is recognized to be largely a function of molecular weight. The values selected are evaluated as reasonable for all dioxin-like compounds, since the molecular weight for these compounds are similar.

8.4. UNCERTAINTIES ASSOCIATED WITH EXPOSURE PATHWAYS

The purpose of this section is to qualitatively describe the uncertainties associated with exposure estimates for the exposure pathways that are included in this methodology. The principal focus is on the exposure parameters - the contact rates and fractions, exposure durations, and so on. A brief summary is also presented on some of the findings pertaining to the

fate, transport, and transfer algorithms used to estimate the exposure media concentrations. This summary will highlight findings that have been included in other sections of this chapter, Chapter 7 on model comparisons and model validations, as well as a section in Chapter 6 on User Considerations. Each section below includes a table summarizing key points of uncertainty. Section 8.4.1 looks at three key exposure parameters which are common among all pathways - lifetime, body weights, and exposure durations. Sections 8.4.2. to 8.4.11 are pathway-by-pathway discussions.

8.4.1. Lifetime, Body Weights, and Exposure Durations

Values for lifetime of 70 years and adult body weight of 70 kg are traditionally used for risk assessment purposes, although data in the Exposure Factors Handbook (EPA, 1997) suggest that the current average body weights may be lower and the lifetime may be longer. The deviations are small and more precise numbers would not change exposure estimates by a meaningful amount. The uncertainty regarding body weight is reduced in the ingestion pathways of fruit/vegetables and the terrestrial animal food products including beef, milk, chicken, and eggs. This is because the consumption rates used in these pathways for the demonstration does in this assessment are in units of g/kg/day and were derived from survey data which incorporated the amount consumed with the individual body weight. Specifically, these rates originated from the household portion of the National Food Consumption Survey conducted by USDA (USDA, 1992). Chapter 2 describes the use of this survey data in detail and Section 8.4.7 below summarizes some of the uncertainties in using it. The assumed child body weight of 17 kg (for ages 2-6) is well founded and not expected to introduce uncertainty into soil ingestion exposure estimates.

Assumptions on exposure durations are the most uncertain of the three parameters discussed here. A value of 9 years assumed for central exposure scenarios was the 50th percentile of time living at one residence derived from census survey data (EPA, 1997). Such mobility surveys typically ask respondents how much time they are living at one residence, so a result such as this one will likely be an underestimate because respondents are likely to continue to live at their residence beyond the time they answered the survey question. The estimate of 30 years for the average residence time of farming families (used to define high end exposure scenarios) was also based on survey data which showed that the 90th percentile time spent in one residence was 32.7 years. For the high end scenarios of this assessment, this 90th percentile is justified based on the definition of high end. Also, however, it is supported based on a

qualitative judgement that farming families may tend to live longer in one spot as compared to non-farming families.

Exposure durations are also tied to assumptions about source strength over time. Assuming 30 years of exposure to stack emissions, for example, assumes that the source of stack emissions will be (or has been) in operation for this length of time with the same stack emission controls in place. The same is noted for the effluent discharge source category. If the source is contaminated soil, assumptions include whether or not the soil will be removed, the site will be capped, and so on. Another consideration is the dissipation of soil residues. Section 8.3 discussed uncertainties with the assumption of non-degradation of dioxin-like compounds in soil when the soil itself is contaminated. A 25-year dissipation half-life is applied to residues which migrate to an exposure site to impact only a thin layer of surface soil. Specifically, a simple soil mixing model incorporating the 25-year dissipation half-life is used to calculate steady state soil concentrations of dioxin in a thin surface layer resulting from atmospheric depositing dioxins, from the stack emission source, or from soil eroding from a nearby site of soil contamination. As discussed above in Section 8.3., an assumption of non-degradation during periods of exposure in the range of 30 years is reasonable, since degadation/dissipation pathways lead to very slow decline of dioxin concentrations in soil.

Exposure estimates are linearly related to all three exposure parameters - increasing body weight and lifetime decreases exposures in an inverse linear fashion, while increasing exposure durations increase estimates in a direct linear fashion.

Uncertainties associated with body weight, lifetime, and exposure durations are summarized in Table 8-1.

8.4.2. Soil Ingestion Exposure

This exposure is directly a function of the concentration of contaminants in surface soil layers. For example Scenarios 1 and 2, demonstrating background conditions, soil concentrations at the site of exposure were set at levels corresponding to an actual setting which can described as, “background”. For example Scenario 3, demonstrating the soil contamination source category, erosion onto the site of exposure deposited residues into a thin, no-till, surface layer of 2 cm, and a thicker, 20-cm, till layer of soil. Soil ingestion exposures were based on concentrations in the 2-cm layer. In Scenarios 4 and 5 demonstrating the stack emission source category, contaminated particles deposited onto the exposure site, also creating a till and a no-till concentration. The no-till depth for this category was also 2 cm.

Discussions on the methodology to estimate exposure site soil concentrations resulting from erosion of contaminated soil from a nearby site are contained in Section 6.3.3.2, Chapter 6, which was on sensitivity analysis and the impact of different parameter values on estimated exposure site soil concentrations, and in Chapter 7, Section 7.3. discussing literature reports of off-site impacts from soil contamination. While off-site impacts were noted in the literature, no data could be found that was directly amenable to comparison with the scenarios of Chapter 5. The closest site for which data was available was the Dow Site in Midland, Michigan. The ratio of soil concentrations of 2,3,7,8-TCDD in areas described as "background" in the 600 ha site to soil concentrations in the contaminated areas was 1/8 to 1/2 as much (depending on how the contaminated area soil concentration was interpreted) as the ratio modeled in the off-site demonstration scenario. This might imply that the model overpredicts off-site soil impacts, except that the "background" areas in the Dow Site appear substantially further away than the 150 meters in the off-site demonstration scenario. Also, data was unavailable to determine the erodibility of soil at the Dow Site. Had this and other site-specific information been available, a more precise test of the off-site soil impact algorithms of this assessment may have been possible. Still, a key finding in the sensitivity analysis exercises was that the erosion algorithms may be overestimating off-site impacts. No information is available on estimating how much of an overestimation may have resulted, and this finding is not a definite conclusion.

If, in fact, an overestimation is occurring, it could be due to a few different factors: 1) an uncertain dissipation rate - increasing it could reduce soil concentrations, 2) assumed depth of mixing for untilled situations - increasing it could also reduce soil concentrations, and 3) the steady state simplification. These factors were examined in the sensitivity analyses conducted in Chapter 6.

In contrast to the possible overprediction of soil concentrations for the soil contamination source category, an exercise described in Chapter 7, Section 7.3.8 suggested that the stack emission source category may be underpredicting soil concentrations. Measured air concentration in an actual rural setting were used in a model validation exercise which attempted to duplicate measured soil concentrations at that same setting. It was seen that modeled soil concentrations were slightly lower than measured soil concentrations. Two possible causes for this underprediction were offered: 1) the model does not account for deposition of vapor-phase dioxins, either through direct deposition or by detritus production, and 2) the representative air profile was derived from samples in March, April, and June, and the average may not have represented typically higher wintertime air concentrations.

In the stack emission source category and the soil contamination source category where the site of exposure is distant from the site of contamination, the two key uncertain parameters are the depth of mixing and the soil half-life for dioxins depositing onto the site of exposure. The mixing depth is a theoretical parameter for which little data is available. The data of Brzuzy and Hites (1995) on soil profiles of dioxins for undisturbed soils does show that dioxins migrate below the surface, in some cases under sandy conditions, to depths greater than 30 cm. However, their non-sandy soil profiles showed most of the dioxins within 5 cm of the surface, and considering that their undisturbed soil cores reflect depositions of dioxins which were speculated to have occurred 50 years or more, the assumption of 2 cm is felt to be reasonably justified. Others have assumed depths of mixing of 1 cm for analogous applications. Evidence from radioactive fallout suggests depths no deeper than 5 cm. Sensitivity analysis on the erosion algorithms showed that assuming a depth of 1 cm instead of 2 cm would have increased soil concentrations by a factor of 2.5, while decreasing the mixing depth to 10 cm decreases soil concentrations by 60%. Very little data is available on dioxin soil half-lives, but the assumption of a half-life of 25 years is within the range of 25-100 years hypothesized by Paustenbach, et al. (1992a) for surface and buried residues based on their survey of the available literature. The analysis by McLachlan, et al. (1996) on data on dioxin concentrations in a plot of soil amended with sewage sludge over 20 years earlier showed half-lives consistently around 20 years for the suite of dioxin congeners, and this is probably the best support for the use of a constant half-life for all dioxin congeners.

Another issue is whether children should be assumed to be exposed to tilled soils - tilled by home gardening, farming, etc. - or untilled soils. It is feasible that children would be exposed to tilled soils in farming or home garden settings. If the soil was impacted by stack emission depositions or erosion from a nearby site of soil contamination, then tilling would reduce soil concentrations. However, it is more reasonable to assume that they generally play outside in areas that are not mechanically tilled.

The estimated soil ingestion quantity is based on field measurements, using trace elements, of soil ingested by relatively small groups of children over brief periods. Methodological issues in these studies remain to be addressed. In particular, ingestion estimates may have been lower if dietary intake of the trace elements was taken into account. Research is underway to refine soil ingestion estimates obtained through trace element measurements. Given the available data, EPA (1997) suggests that 100 mg/day is a reasonable central estimate for children under 6 years of age, and that value is used in this assessment in the central scenarios. Due to the behavior known as pica, some children are known to ingest high amounts of various

non-food materials. Estimates of pica ingestion of soil by children have ranged as high as 5000 mg/day. The high end estimate of 600 mg/day is not characterized as pica. It was determined from studies evaluated in EPA (1997) which showed upper percentile estimates ranging from 106 mg/day to 1,432 mg/day with an average of 587 mg/day for soil and dust ingestion.

Soil ingestion exposure estimates also depend on the duration of the period over which children are assumed to ingest soil. Data on soil ingestion by age are not available, and the estimate that significant ingestion occurs between ages 2 and 6 is broadly supportable on behavioral grounds.

No measurement data are available on soil ingestion in infants (0-2 yrs. old) or in older children or adults, and no ingestion is assumed for these groups. While some soil ingestion will occur in these groups, e.g., through contact of soiled hands with food, it is plausible that such ingestion is of a lesser degree than occurs in early childhood. If Hawley's (1985) estimate that an adult ingests an average 60 mg/d of soil is used, after accounting for differences in exposure duration (9-20 yrs versus 5 yr) and body weight (70 kg versus 17 kg), the adult soil ingestion exposure is close to the estimated exposure for children (at 200 mg/d). The high end example scenarios in Chapter 5 assumed that the exposed family was involved in farming operations. One implication is that individuals on the farm would be working closely with the soil, which may result in some soil or dust ingestion (dust ingestion is distinct from the particulate inhalation exposure pathway). The other implication is that, should this be the case, they might be in contact with tilled or otherwise well mixed soil, whose concentration could be as much as 10 times less than the no-till soil for which children are assumed to be exposed.

Considering these uncertainties, the soil ingestion exposure estimates presented for children are plausible. Further consideration may be warranted for considering adult soil ingestion, particularly in farming situations. Uncertainties associated with the soil ingestion pathway are summarized in Table 8-2.

8.4.3. Soil Dermal Contact Pathway

Estimates of dermal exposure to soil rely largely on four factors unique to this pathway: exposed skin area, soil adherence (also termed soil contact), frequency of soil contact and fraction of contaminant absorbed. The uncertainty in these three terms are discussed below.

Before that discussion, a brief note is made on uncertainties associated with soil concentrations. Discussions above on the soil ingestion pathway addressed uncertainties associated with soil concentrations which result from migration of residues from a distant source to the site of exposure. Distant sources in this assessment include off-site soil contamination and

stack emissions. Discussions in the soil ingestion pathway section above pertain to this exposure pathway and are not repeated here. However, there is one key difference in the soil dermal and soil ingestion pathways. Soil ingestion exposures are assumed to occur only from surficial soil layers and from untilled soils, which translates to the 2-cm mixing depth for both the "central" (residential) and "high end" (farming properties) scenarios. Soil dermal contact, on the other hand, is assumed to occur in association with both tilled and untilled soils . "Indoor" soil is assumed to have concentrations equal to that of untilled soils, while "outdoor" dermal contact events are assumed to occur in association with gardening or farming activities, where the concentrations are the more dilute tilled concentrations.

The range of possible estimates of exposure via dermal contact is probably more a function of variability in the population than uncertainty in the dermal contact methodology and assignment of exposure parameters. Relatively accurate measurements have yielded a good data base on total skin area. Thus the uncertainty in this factor is derived more from the assumptions of how much of the total skin area is exposed. EPA (1992b) recommends approaching this issue by determining the coverage of normal apparel in the exposed population and assuming exposure is limited to the uncovered skin. As discussed in EPA (1992b), this assumption could lead to underestimates of exposure since studies have shown that some exposure can occur under clothing, especially in the case of vapors or fine particulates. Assignment of skin surface areas in this assessment have assumed estimates for various combination of areas for hands, arms, and legs. The extent to which individuals where short or long sleeve shirts and trousers is part of the variability in skin surface area assignment.

The potential for soil contact and subsequent adherence probably varies little across the population, but few actual measurements have been made. A wide range of from <0.002 to >20 mg/cm^2-event has been identified in EPA (1997). The very high adherence rates were found for the scenario described as, "kids-in-mud", and was from data on children playing by a lakeshore. The lower range was found for an indoor Tae Kwon Do setting. Adherences for a day-care setting ranged from 0.03 for arms and legs to 0.1 for hands. Outdoor adherences for gardeners ranged from 0.005 for legs to 0.02 for arms to 0.2 for hands. The uncertainty in these estimates reflect primarily the lack of measurement data rather than population variability. Site variability is probably important as well since soil properties such as moisture content, clay content and particle size distribution are likely to affect adherence.

Exposure frequency to soil reflects largely personal habits and thus the range in values for this parameter is primarily based on population variability. Seasonal and climate conditions can also affect this behavior introducing site variability as well. Indoor contact events were assumed

to occur daily, gardening events were assumed to occur 100 times per year and farming events 350 times per year. These values were assigned based on judgement, and not any particular studies.

The dermal absorption fraction of compounds varies widely across chemicals, whereas skin properties that affect absorption, i.e. thickness and composition vary little across the population. Thus the uncertainty in this factor is derived primarily from measurement error rather than population variability. Soil properties, such as organic carbon content, can also affect the extent of dermal absorption and thus create site variability as well. EPA (1992b) reports two studies which measured dermal absorption of 2,3,7,8-TCDD from soil. Testing included human skin in vitro, rat skin in vitro and rat skin in vivo. On the basis of these tests, a range of 0.1 - 3.0% was recommended in EPA (1992b). Dermal absorption testing, especially for soils, is a relatively new field and many uncertainty issues are involved. These include extrapolation of animal tests to humans, extrapolation of in vitro to in vivo conditions, and extrapolation of experimental conditions to expected exposure conditions. Extrapolation of the tests on 2,3,7,8-TCDD to the other dioxin like compounds (which have not been tested) introduces further uncertainties. A dermal absorption fraction of 3.0% was adopted here for application to all the dioxin like compounds. Based on the observed range of values for 2,3,7,8-TCDD this assumption may lead to overestimates of a factor of 30. Considering all possible uncertainties, under estimates are also possible, though judged less likely.

In summary, dermal exposure estimations rely on a number of parameters whose values are not well established. The range of possible dermal contact estimations is judged to be mainly a function of population variability, rather than parameter uncertainty. One parameter that is uncertain is the absorption fraction. The value selected for this assessment. 0.03 (3% absorption) is on the upper end of the range of suggested values, so its selection is likely to result in overestimating, rather than underestimating, the exposure due to this pathway. Although it is difficult to estimate the overall variability and uncertainty with this pathway, it is judged to be plus or minus one to two orders of magnitude. A summary of the uncertainties associated with the dermal absorption pathway is given in Table 8-3.

8.4.4 Water Ingestion

The strong sorptive tendencies of the dioxin-like compounds result in very low water concentrations. Monitoring for CDD/Fs mostly have not found these compounds at a detection limit around 1 pg/L (ppq), and when found, have generally been very near this concentration. The one exception is an upstate New York community water system, where tetra through octa-

CDFs were found at concentrations ranging from 2 pg/L (tetra) to over 200 pg/L (octa). The surface water concentrations predicted by the algorithms of this assessment for all source categories are 10^{-2} pg/L and lower, which is consistent with the sparse monitoring data. Although there was no data found that could be directly applicable to the source categories, it does not appear that the models estimating water concentrations will introduce significant uncertainty into water ingestion exposure estimates.

The classically assumed water ingestion rate of 2.0 L/day was examined in EPA (1997). The conclusion was that this estimate is more appropriately described as an upper percentile consumption rate for adults, and recommended 1.4 L/day for use as an average. This value was used for water ingestion in the central scenarios. EPA (1997) cautions that data on consumption rate for sensitive subpopulations such as manual laborers are unavailable. As such, the 1.4 L/day rate for individuals in farming families who work the field may be low. For this reason, a 2.0 L/day was assumed in the high end, farming, scenarios.

The contact fraction is defined as the fraction of total contact with an exposure media that is contact with contaminated media. For drinking water, this translates to the fraction of water ingestion that comes from the contaminated water source. In the example scenarios, it was assumed that the impacted water was a river which supplied water to the exposed individuals, perhaps through a public water system. The contact fraction of 0.70 for central scenarios is based on time use surveys which showed roughly this fraction of time spent in and around the home environment on the average (EPA, 1997). The upper limit is, by definition, 1.00; this was felt to be unrealistic even for high end scenarios. EPA (1997) recommends an upper end value for time at residence at 0.90, and this value was used for the high end scenarios.

The uncertainties associated with the water ingestion pathway are summarized in Table 8-4.

8.4.5. Fish Ingestion Exposure

Chapter 7, Sections 7.3.5 and 7.3.6 addressed the capabilities of the models of this assessment to estimate fish tissue concentrations, by comparing measured fish concentrations with modeled concentrations. In general, it was concluded that modeled fish tissue concentrations in background settings are consistent with those found in the literature for similar settings. Also, impacts of point source discharges into surface water appear to have been appropriately modeled.

Chapter 7, Section 7.3.2. looked at a comprehensive data set developed and supplied by the Connecticut Department of Environmental Protection which included soil concentrations,

sediment concentrations of water bodies near where soil samples were taken, and fish concentrations from the same water bodies. Data on 2,3,7,8-TCDD, 2,3,7,8-TCDF, 2,3,4,7,8-PCDF, and total TEQ were examined. Soil concentrations of 2,3,7,8-TCDD were found to be in the low ppt range, which has been described in various places in this document as a range for "background" soil conditions. Sediment concentrations of the three congeners and total TEQ were generally in range of 2-3 times higher than soil concentrations, which was consistent with the demonstration of background conditions. This demonstration scenario had a basin-wide 2,3,7,8-TCDD soil concentration of 0.37 ppt, and the sediment concentration was estimated at 0.99 ppt. The Biota Sediment Accumulation Factor, BSAF, from this field data was estimated to be 0.86 for 2,3,7,8-TCDD. This was higher than the assumed 0.09 in the demonstration scenarios. Two explanations were offered for this difference. One was that the fish sampled were bottom feeders, which would put them in more contact with contaminated sediments compared to column feeders, and the 0.09 value was based on data from column feeders; higher impact from contaminated sediments is expected from bottom feeders as compared to column feeders. Two, the 0.86 may have been skewed from two (of seven) sites in the Connecticut data which had high BSAFs at greater than 1 and 3. Although the soil sampling in this data set was generally sparse, the result that bottom sediment concentrations exceeded surface soil concentrations by 1.6-3.9 times generally supports the model's algorithms for estimating sediment concentrations in areas with low basin-wide concentrations.

Chapter 7, Section 7.3.5 looked at fish concentrations in background areas and where point source impacts to water bodies were identified. A principal source of information was EPA's National Study of Chemical Residues in Fish (EPA, 1992a; abbreviated NSCRF). The range of fish tissue concentrations of 2,3,7,8-TCDD measured for (perhaps) background conditions in this study, 0.56 - 1.02 ppt, were comparable to the fish tissue concentration estimated assuming the low (perhaps) background soil concentration of 0.37 ppt soil concentration, 0.2 ppt. It may not be appropriate, however, to make the same observation for the source categories assuming higher soil concentrations as compared to measured concentrations. In this case, the range of measured concentrations, 1.4 - 30.02 ppt, does not compare with the modeled 0.3 ppt. It was noted that the soil contamination source category was demonstrated with a setting that had four hectares of contaminated soil at 1 ppb surrounded by a watershed of 100,000 hectares with a 0.0 soil concentration, which may explain partly why the results did not compare with the concentrations in the NSCRF that were taken near contaminated sites. Specific field data were not available for more detailed analysis. In general, it would appear that the magnitude of concentrations appears to have been captured for background situations.

While the modeled PCDD/PCDF fish concentrations seem reasonably in line with measured concentrations, this assessment may have underestimated concentrations of 2,3,3',4,4',5,5'-HPCB in the demonstration scenarios. Concentrations for fish in the Great Lakes Region were in the tens to hundreds of ppb range, while this assessment derived estimates all under 1 ppb. However, an examination of bottom sediment concentrations of PCBs in the literature showed them to be roughly three orders of magnitude higher than estimated with the algorithms of this assessment. This mirrors the difference in observed versus estimated fish tissue concentrations. The Biota Sediment Accumulation Factors, BSAFs, for PCBs also was noted to be variable, with values below 1.0 to values over 20.0 (see Chapter 4, Section 4.3.4.1). The BSAF for the example PCB congener in this assessment was 2.0. Higher BSAFs would also increase PCB concentrations estimated for fish.

Chapter 7, Section 7.3.6 evaluated the model for estimating fish tissue concentrations for the effluent discharge source category, using data from the 104-mill study. Comparing model predictions of fish tissue concentrations with observed concentrations, it was found that there was generally an underprediction of observed fish tissue concentrations, although the average predicted concentration 7 ppt cannot be considered significantly different then the observed average concentration of 15 ppt. An important qualifier is that this exercise assumed that the effluent discharges were the sole source of contaminants which may have impacted the water bodies. Also, the maximum "observed" fish tissue concentration of 143 ppt was matched by a predicted concentration of 89 ppt, which was also the maximum predicted concentration. Finally, there was discussion that the BSSAF (biota suspended sediment accumulation factor) assigned value of 0.09 for 2,3,7,8-TCDD, the same value used for the BSAF, might be low for the effluent discharge source category. The justification for this hypothesis concerns the differences between past and ongoing water body impacts, and the fact that the 0.09 value was based on field data for a water body where impacts are speculated as principally occurring in the past (see Section 7.2.3.6 for a further discussion of this issue). When the BSSAF was "calibrated" to 0.20, the average predicted fish concentration of 15 ppt for 2,3,7,8-TCDD now matched the observed average fish tissue concentration.

The model did not perform as well for pulp and paper mills discharging into the largest receiving water bodies. The average fish tissue concentration observed for 21 fish was about 7 times higher than predicted concentration. No precise conclusion can be reached with this result, although modeling lower fish concentrations in a large receiving water body than are measured does not appear unexpected. Large water bodies are likely to be ones having multiple sources of dioxin release in comparison with small water bodies. Therefore, the assumption that one or

more proximate mills are solely responsible for observed fish concentrations is most likely to be flawed for large water bodies.

In summary, the evaluations for model performance regarding fish tissue concentration estimation seem to lend credibility to the approaches taken, despite the simplicity of the of dilution models chosen. The sensitivity analyses exercises on the algorithms to estimate fish tissue concentration discussed the variability and uncertainty with the parameters required for the algorithms. Generally, the most sensitive input was the source strength characteristics - soil concentrations, contaminant discharge rates in effluents, and so on. A single order of magnitude or less range in predicted concentrations would result with singular changes in all other model parameters.

An exposure parameter of paramount importance in estimating exposure to contaminated fish is the fish ingestion rate. Available fish consumption surveys are discussed in EPA (1997). They were divided into five subsets of surveys, one of which was titled, "freshwater recreational anglers". Three surveys in this subset were deemed appropriate for generation of consumption rates, and EPA (1997) recommended a mean and a 95^{th} percentile consumption rates of recreationally caught fish of 8 and 25 g/day, respectively. Another possible approach is described in EPA (1989) and was used in a previous version of this dioxin reassessment document (EPA, 1994). Briefly, this approach assumes a meal size and then determines, on a site-specific basis, the number of meals an individual would consume from fish obtained from the impacted water body. EPA (1994) assumed meal sizes of 150 g/meal, and 3 and 10 meals/year for the central and high end assumptions, respectively, which led to daily consumption rates of 1.2 and 4.1 g/day. Assessors should also be cognizant of situations where subsistence fishing can lead to much higher rates of fish consumption. EPA (1997) summarizes studies where subsistence patterns of fish consumption can lead to consumption rates in the hundreds of grams per day. Like other food consumption pathways, which have the highest exposure estimates for dioxin-like compounds, obtaining site-specific information for fish ingestion is critical for this pathway.

A summary of the uncertainties associated with the fish ingestion pathway is given in Table 8-5.

8.4.6. Vapor and Particle Phase Inhalation Exposures

This section will address the uncertainty associated with vapor and particulate phase inhalation exposures. Sources addressed in this assessment include stack emissions and contaminated soils; this section will only address contaminated soils. The fate and transport of

dioxin-like compounds from stack emissions to exposure sites, and the resulting air concentrations, are discussed in Chapter 3.

The respiration rates of 13 and 20 m^3/day used for inhalation exposures is based on data described in EPA (1997). The contact fraction is 0.70 for central scenarios and 0.90 for high end scenarios. Like the water ingestion contact fractions, these were based on time at home surveys. The inhalation rate and contact fractions are not expected to introduce much uncertainty into inhalation exposure estimates.

Another exposure parameter critical for the inhalation pathway is exposure durations, which is 9 years for central and 30 years for high end exposures. The uncertainties associated with this parameter in its use as an exposure parameter are discussed above in Section 8.4.1. However, exposure duration is additionally critical for the inhalation pathway for the soil contamination source category, as estimated volatilization flux is a function of the time during which volatilization is occurring. Essentially, the model assumes that contamination is at the soil surface at time zero, and over time, residues which volatilize originate from deeper in the profile leading to lower volatilization fluxes after time, and also lower average volatilization flux as the averaging time increases. The sensitivity analyses exercises in Chapter 6, Section 6.3.3.1., evaluated the sensitivity of air concentration predictions to changes in exposure duration. It was shown that there is roughly a factor of four difference between concentrations predicted over one year duration to a seventy year duration. Therefore, there is both a direct and an indirect impact from changing the exposure duration in these procedures. The direct impact from changing exposure duration is in the exposure equation - increasing the exposure duration increases the exposure estimate. What is seen also with increases in exposure, however, is a decrease in the estimated average air concentrations to which individuals are exposed. The impact in the exposure estimates is more driven by having more years of exposure rather than being exposed to a lower average air concentration, as expected.

Vapor-phase emissions from soils are estimated with a volatilization flux algorithm. The procedures were developed in Hwang, et al. (1986). A near-field dispersion model estimates air concentrations for the circumstance where the soil contamination is at the site of exposure. Where the site of contamination is located distant from the site of exposure, the same volatilization flux model is used, but exposure site concentrations for these sources are estimated using a far-field dispersion model.

Sensitivity analyses in Chapter 6 showed that the air concentration varied roughly over an order of magnitude with testing of key contaminant parameters, the organic carbon partition coefficient, Koc, and the Henry's Constant, H. Air concentration predictions are also sensitive to

other key parameters, including those associated with source strength (area of contamination, concentration), geometry, (distance to receptor in off-site source category), and climate (average windspeed). However, these might be expected to be known with a reasonable degree of certainty for a site-specific application. If they are, it can be concluded that the most uncertainty associated with the vapor phase algorithm is in the contaminant parameters, and it would appear that a range of about an order of magnitude difference in predicted air concentrations might be expected with different pairs of these parameters.

A model validation exercise described in Chapter 7, Section 7.3.8 tested the algorithms modeling air concentrations above a soil of known concentrations. Using measured soil concentrations at a site near Columbus, Ohio, and measured air concentrations at this same site, it was shown that the model predictions of air concentrations were orders of magnitude lower than measured air concentrations. While this suggests that the model is underpredicting the release of dioxins from soil into the air and/or underpredicting the dispersion of released residues, it may be true, on the other hand, that the measured air concentrations in the rural setting near Columbus are the result of long range transport of air-borne dioxins from distant sources of release.

Another piece of evidence came in an examination of above ground plant:soil ratios as generated by the models and found in experimental testing. The models underestimated these ratios by 1 to 2 orders of magnitude as compared to the literature when vegetation in the field studies described in the literature were grown in soils with concentrations in the ppt range, a range typical of background settings. Two explanations were offered for this trend: the experiments were impacted by sources of dioxins other than the soil in which the plant was growing, and/or, the soil-to-air models may be underestimating air concentrations. Like the model validation exercise described above, it is unclear which explanation dominates the observed trend.

An alternate model for volatilization flux and an alternate model for air dispersion were evaluated in Chapter 7, Section 7.2.4. It was found that the alternate volatilization model predicted about a third as much volatilization as the Hwang model, but that the alternate dispersion model predicted air concentrations that may be up to an order of magnitude higher than the models predicted in this assessment.

There was no data on concentrations of air-borne contaminants in the particle phase only. The procedures used to estimate the suspension of particles were developed from information on highly erodible soils. As such, fluxes and hence concentrations may be higher than expected. However, with no data to compare, this cannot be ascertained. It was seen that vapor phase concentrations exceeded particle phase concentrations by over an order of magnitude. The

sensitivity analysis exercises in Chapter 6 did indicate a two order of magnitude range in estimated concentrations depending on the assumptions concerning wind erodibility of the soil. Also, several issues of uncertainty concerning the suspension of contaminated particles and relationship between air-borne vapor and particle phases were examined. It was noted that the total reservoir of suspended contaminated particulates was likely to be underestimated because the algorithm for wind erosion was developed only for inhalable size, < 10 μm, particles, which is appropriate for inhalation exposures but would lead to an underestimate of the depositions onto vegetation, including fruits/vegetables for consumption and grass/feed for the beef/milk bioconcentration algorithm. Vegetation concentrations might also be low because the impact of rainsplash on transferring soil to the lower parts of vegetation was not considered.

A critical assumption made was that volatilized residues remained in the vapor phase and did not sorb to airborne particles. This led to a dominance of vapor phase contaminants - 90% and more of the total airborne reservoirs (vapor + particle phases) estimated for the on-site and off-site soil source categories were in the vapor phase. Even though only three contaminants were modeled for the soil source category, this trend would be repeated for essentially all the dioxins (except not as much for the octa dioxins since the models would predict much less vapor phase release than the other dioxins). Having much more vapor phase dioxins than particle phase dioxins is inconsistent with the vapor/particle partitioning models used to partition ambient air dioxins into vapor and particle phases, and also inconsistent with monitored vapor/particle partitioning. For example, the vapor/particle partitioning model resulted in a prediction that 51% of the total airborne 2,3,7,8-TCDD would exist in the vapor phase, not over 90%. For the other dioxins, the particle phase is predicted to dominate the air concentrations. This suggests that the soil models of this assessment are deficient in that they do not repartition soil emitted dioxins. Specifically, a portion of the vapor-emitted dioxins are unlikely to remain as vapor, but are likely to sorb to particles. Transferring portions of the vapor phase contaminants to the particulate reservoir to get balances suggested by the vapor/particle partitioning models of this assessment would not change total inhalation exposures, but would impact concentrations in above ground vegetation. Currently and even with transfers such as these, vapor phase transfers dominate plant concentrations. Because vapor phase reservoirs would be reduced after transferring a portion to the particle phase, such transfers translate to reductions in plant concentrations, and for grass and feed, subsequent reductions in beef and milk concentrations and exposure estimates.

Perhaps the most critical assumption which could be questioned is that airborne vapor and particle phase contaminants at the site of exposure originate only from the site of contamination when the site of contamination is distant from the site of exposure. Meanwhile,

soils at the exposure site are impacted - concentrations in the air at the exposure site do not consider possible fluxes from exposure site soils, or from soils between the contaminated and exposure sites.

A test was conducted for this assumption using the demonstration scenario for the soil contamination source category, which had a 4-ha site at 1 ppb 2,3,7,8-TCDD 150 meters from an exposure site of the same size. The soil concentrations at the exposure site were 0.36 ppb for a 2-cm notill mixing depth and 0.06 ppb for a 20-cm tilled mixing depth. These concentrations were then input as soil concentrations for the soil contamination source algorithms to determine what air concentrations would result above the soil. For this test, the "near field" dispersion algorithms described in Chapter 4 were used instead of the "far field" algorithms used in the demonstration of that source category in Chapter 5. These near field exposure site air concentrations, generated with a starting soil concentration of 0.36 ppb, were compared with exposure site air concentrations generated when using the far field dispersion algorithms, starting with the soil concentration of 1 ppb. It was found that on-site air concentrations with soil concentrations at 0.36 ppb exceeded exposure site vapor and particle air concentrations estimated for a 1 ppb contaminated site 150 meters away by a factor of about 5. When the same test was run using a tilled concentration of 0.06 ppb, concentrations predicted using the near field algorithms and this concentration were similar to the concentrations predicted using far field algorithms and a starting concentration of 1 ppb.

Several uncertainties were discussed, but a lack of data and a complete understanding of atmospheric processes for dioxin-like compounds precludes any final quantitative judgements on uncertainties in the air concentration algorithms. Some of the uncertainties imply that procedures and assumptions adopted overestimate pertinent environmental media, and others imply that such media concentrations were underestimated. The assumption that air-borne reservoirs of contaminant originate only at an off-site area of contamination and not from other soils should be examined further.

A summary of the uncertainties associated with the vapor and particle inhalation routes is given in Table 8-6.

8.4.7. Fruit and Vegetable Ingestion

Consumption rates of 1.49, 1.52, and 1.16 g/kg/day were derived in EPA (1997) from the household portion of the National Food Consumption Survey (NFCS; USDA, 1992). Contact fractions of 0.101 for fruits and 0.173 for vegetables were also obtained from an analysis of NFCS data. Briefly, the household portion of the NFCS was a survey filled out by the head of a

household and includes the amount of food product brought into the house for consumption. The data includes the number, age, and weight of all household members, in addition to critical questions concerning home production of foods. Further details on this survey and the use of the data is described in Chapter 2.

Use of this portion of the NFCS has its benefits and drawbacks. One major benefit is that it reduces uncertainty by the calculation of rates which include body weights. The earlier version of the dioxin reassessment document (EPA, 1994) had the consumption of all food items in terms of g/day. In this assessment, fish ingestion is still handled this way, but all other foods considered (vegetables/fruits, terrestrial animal food products) more appropriately consider the interaction between rate of consumption and body weight. Another major benefit is that it allows one to estimate how much of a food product is consumed in a household which was produced by the household, which is precisely what is desired for the demonstration scenarios of this assessment. This estimation includes a reported consumption rate and also a contact fraction ascertained from survey data. Another part of the NFCS was called the "1-day individual consumption survey". One cannot ascertain consumption rates for home-produced foods from the 1-day survey. However, the individual survey does ascertain the consumption rate for foods "as eaten" by the individual. In contrast, the household survey asked for total food product brought into the house for consumption that week. That necessitates assumptions on the meal size per individual in the household, and in addition to data on the weight of the household individuals, EPA (1997) derived estimates of g/kg/day consumption rates, which were used in this assessment. That also necessitates a consideration of the amount of the total food product brought into the house which is not eaten by individuals in the house, since the "total food product" is not a quantity analogous to, "as eaten". Reductions in this total would include losses such as from cooking, discarding part of the food product, such as bones or uneaten portions, or portions given to guests. This is one disadvantage to the household survey, in contrast to the "as eaten" data from the 1-day consumption survey. In Chapter 2, reduction factors are described and used in this assessment to describe cooking (weight reduction) and post cooking (bones, etc) for beef and chicken, as well as other meats not considered in this assessment.

EPA (1997) also ascertained, from questions on specific fruits/vegetables from the household survey, consumption rates for "exposed above ground vegetables/fruits" and "root vegetables". Protected vegetables/fruits, as opposed to exposed, were defined as vegetables/fruits which have outer protective coverings which are removed prior to consumption such as peas or oranges. No root vegetables were considered to be protected although, of course, it is common to consume some below ground vegetables such as carrots or potatoes after removal

of the skin. Consumption rates for exposed fruits and vegetables are desired because the evidence is fairly clear that dioxin-like compounds will not penetrate through thick skins which are peeled prior to consumption. This assessment does consider, however, the peeling of skins off exposed vegetables. This consideration is in the form of a "VG" parameter. The VG parameter, which includes separate assignments for above ground vegetables/fruits, VG_{ag}, and for below ground vegetables, VG_{bg}, considers the following: evidence that little translocation from the surface of bulky vegetation, below or above ground, to the inner portions of these vegetation, and any additional consideration of the peeling of the skin (carrots, potatoes, e.g) prior to consumption. In this assessment, values of 0.01 and 0.25 were assigned to VG_{ag} and VG_{bg}, respectively.

All these assumptions discussed: total consumption rates, protected or unprotected, above or below ground, and fraction home grown, are probably reasonable for general assessment purposes as long as exposures are to the broad categories of fruits or vegetables, and not for individual fruits or vegetables. For a site specific assessment, there will likely be wide variability on the types of produce grown at home, what percentage of that is unprotected, and so on. Finally, and as is also true for beef and milk exposures, this assessment only considers the impact of home-grown fruits and vegetables. In rural settings, it is plausible that a large percentage of an individual's total fruit and vegetable intake comes from nearby and impacted sources, more than the 10-20% assumed in this assessment. If all of the consumption of fruit and vegetables is from local sources, and adjustments are made to correctly predict concentrations in local fruits and vegetables, than contact fractions should be set at 1.0, and exposures could increase up to 10 times compared to the demonstration scenarios depending on concentration estimation.

Several issues of uncertainty pertinent to the estimation of concentrations in below and above ground vegetation have been examined in other parts of this document and are not repeated here. Key issues include: 1) the uncertainty associated with empirical parameters, VG_{ag} and VG_{bg}, 2) the assumption that residues which volatilize from contaminated soils remain in the vapor phase and not partially partition into the vapor phase, 3) the possible underestimation of total particle reservoirs of contaminant in the air resulting from wind erosion of contaminated soils because the wind erosion algorithm only estimated suspension of inhalable size and not all particulates, and also because the possible effect of rainsplash onto vegetables low to the ground such as lettuce, was not considered, 4) for the stack emission source, uncertainties associated with air dispersion and deposition modeling using the ISCST3 model as discussed earlier in Section 8.2, and therefore the subsequent impacts of air-to-plant and soil-to-plant transfers, 5) for the stack emission and off-site soil source categories, air borne concentrations in the vapor and

particle phases at the exposure site are assumed to only originate at the source of contamination (the off-site contaminated soil and stack emissions) and not on impacted soil at the exposure site - considering additional fluxes from impacted soils other than exposure site soils could lead to up to an order of magnitude higher concentrations in the vapor and particle phases, which in turn affect above ground vegetation, and 6) also for the stack emission and off-site soil source categories where garden soil concentrations are predicted and then used to predict concentrations in underground vegetables, there are uncertainties for the soil concentration algorithm, particularly in the assignment of half-life, mixing depths, and lack of consideration of detritus production and vapor impacts to soils.

Quantitative judgements as the uncertainties associated with these issues are difficult to make. An examination of experimental data in Chapter 7, Section 7.3.10, where most of the vegetation were grown in well characterized conditions implied that the soil contamination models may be underestimating concentrations in above ground vegetables. The evidence examined was plant:soil ratios for experimental conditions versus what the models would predict. This could be due to underestimation of air concentrations of dioxins originating from soils, and there was some suggestion of that. However, it could also be due to the fact that the residues affecting the plants in the experiments were not only from soil releases but from other sources leading to air-borne residues. The models of this assessment only consider air concentrations from the source in question. Therefore, it is hard to ascertain whether the models underpredict, overpredict, or adequately predict above ground vegetation concentrations resulting from soil contamination.

Chapter 7, Section 7.2.1 did look at three modeling approaches for the air-to-plant pathway for leafy vegetation (grass, in particular), including the EPA model. That did section did suggest that the EPA model led to reasonable matches between predictions and observations, with just about a factor of two separating predictions and observations. Also, the air-to-beef model exercise described in Chapter 7, Section 7.3.12 also included an examination of the capability of the air-to-plant model. Although there wasn't a good data set for validation - i.e., measured air concentrations above measured grass concentrations, the examination in that section did support the model's algorithms. In the same vein, it is noted that the vapor phase air-to-leaf transfer algorithm was developed from actual field data. By definition, therefore, it would appear that the air-to-plant modeling are going to predict reasonable plant concentrations. This discussion is put forth only to suggest that the air-to-plant modeling would not be an issue for uncertainty regarding the impact of contaminated soils on above ground plants.

A summary of uncertainties associated with the fruit and vegetable ingestion exposure pathway is provided in Table 8-7.

8.4.8. Ingestion of Terrestrial Animal Food Products Including Beef, Milk, Chicken, and Eggs

The algorithms for the calculation of dioxin concentrations in all these animal food products is the same: they are a function of the weighted average concentration in the diet of the cattle (dairy or beef) and chicken, which is a function of the proportion of the diet in soil and animal vegetation, and a bioconcentration factor. Therefore, previous sections on soil contamination, soil transport algorithms, and plant concentration estimation, are relevant to estimating terrestrial animal food concentrations.

The most critical and uncertain parameters in these algorithms are the bioconcentration factors. The multiplication of the weighted average dietary concentrations of the chicken and cattle by the bioconcentration factors yields a product fat concentration (beef fat, chicken meat fat, milk fat, and egg fat). A set of bioconcentration factors were developed from laboratory feeding experiments for chicken meat, specifically from data on chicken thighs, and a separate set from eggs. There is uncertainty in applying these laboratory derived BCFs to field situations. Data is being developed by these same researchers from chickens which are raised in the field. One purpose of these additional experiments is to verify the laboratory derived BCFs (M. Petreas, Department of Toxic Substances Control, California EPA, personal communication). The bioconcentration factors used for calculation of beef and milk fat were less certain. They were derived from one experiment on one cow and on milk. Besides the sparsity of data, there is obviously uncertainty in applying bioconcentration factors developed from milk fat to beef fat. However, researchers have noted that the dioxin concentrations in beef and milk fat tend to be similar, and this they attribute to the fact that most cattle are slaughtered within 2 years of life while they are still growing. Therefore, the body fat pool is expanding which provides dilution to dioxins taken in by the beef cattle, and as a result, body fat concentrations are found to be similar to milk fat concentrations.

What also strengthens the use of the milk fat BCF to beef fat is the air-to-beef model validation exercise described in Chapter 7, Section 7.3.12. That section described a validation exercise where air concentrations of dioxin-like compounds were routed through the food chain model to estimate concentrations in beef. Generally, that section showed that an air concentration of 0.019 pg I-TEQ/m^3, speculated to be an appropriate air concentration for rural environments where cattle are raised for beef, translates to a beef fat I-TEQ concentration of 0.61

ppt, using the models and parameters of this assessment. The observed beef fat concentration of 0.89 ppt I-TEQ (assuming non-detects were equal to ½ detection limit) was the average from a national, statistically design, monitoring study of dioxins in beef back fat conducted jointly by EPA and USDA. Besides a very reasonable match between observed and predicted I-TEQ beef fat concentration, Section 7.3.12 also describes a reasonable match in the concentrations of the individual congeners.

Other than the critical bioconcentration factors, there is uncertainty with the soil bioavailability factor, B_s, and the parameters describing the chicken and cattle diet which include dietary fractions in soil, grass, and feed (the sum of the three adding to 1.00). The B_s was assigned a value of 0.65 for the beef and chicken algorithms, and reflects an assumption that dioxins are less bioavailable to the animals when the vehicles are soil rather than vegetative feeds. This is a critical assumption for chickens, particularly, since the algorithm for free range chicken impact assumes 10% of the diet in soil, and no exposure through their other diet. This was based on analysis of the chicken feed showing non-detects for dioxins at low detection limits done by the researchers who developed the BCFs, who also developed the rationale for the 10% soil assumption (Stephens, et al., 1995b). The beef cattle diet differs from the dairy cattle diet in that the beef cattle diet is dominated by leafy vegetation (i.e., pasture grass) and partially protected vegetation (a combination category which would include barn feeds such as hay or silage), with 8% in soil. The dairy cattle diet is assumed to be dominated by grains, which are assumed to be protected and residue-free. Only 10% of the dairy cattle diet is assumed to originate from soil (4%) and leafy vegetation (6%).

Section 6.2.3., Chapter 6, described the results of sensitivity analysis of these parameters applied to the beef and milk algorithms. It was shown that there is a small range of possible values for B_s and a small impact on results, for beef and milk at least. The impact, as noted above, would be greater for chickens. Data indicates that range of values for BCF for 2,3,7,8-TCDD is 1 to 10, with a concurrent order of magnitude difference between the upper and lower values. The parameters describing cattle exposure to soils and vegetation at the site are also critical, with up to an order of magnitude difference in concentrations for the example exposure situations examined in Section 6.2.3. It is expected that cattle exposure assumptions can be reasonably described for a specific site. Therefore, the most uncertainty in the bioconcentration algorithm itself lies with the bioconcentration factor, BCF.

Besides the air-to-beef model validation exercise, there was one other literature comparison that was made was comparing beef fat:soil and milk fat:soil concentration ratios derived for PBBs with those estimated for 2,3,7,8-TCDD in the soil contamination demonstration

scenario. Such a comparison is thought to be valid since PBBs are similar in fate and bioconcentration tendencies to the dioxin-like compounds. The field data was from an experiment where the cattle were raised in soils very high in PBB concentration. This provided some evaluation of the beef bioconcentration algorithm as applied to soil contamination. In this comparison, differences in beef and milk bioconcentration tendencies appear to be captured. Fries (1985) found body fat:soil PPB and milk fat:soil PBB concentration ratios for dairy heifers to range from 0.10 to 0.37, and from 0.02 and 0.06, respectively. For body fat of beef cows, these ratios were 0.27 and 0.39. Analogous ratios were derived for the contaminated soil scenario, and for beef and milk fat. For the contaminated soil demonstration scenario, Scenarios 3, beef fat:soil and milk fat:soil ratios were 0.15 and 0.08, respectively. These appear a bit lower than the PBB ratios derived by Fries (1985). The interpretation of this result was that, again here was some evidence that models may be underestimating the impacts of soil contamination to air, and hence air to plants and plants to animals.

Chapter 7, Section 7.2.6 evaluated other beef and milk bioconcentration models; none were found for chickens. It was found that most efforts are quite similar to the model of this assessment, with simple mathematical transformations. Other efforts had considered cattle inhalation exposures and cattle ingestion of impacted water, and found them to be of minimal importance in estimating beef and milk concentrations. They were not considered in this assessment. Two efforts, that of Stevens and Gerbec (1988) and Fries and Paustenbach (1990), evaluated the practice of placing beef cattle on a grain-only diet for fattening prior to slaughter. Both assumed that the reduction in beef concentrations could be modeled as a first-order process with a half-life of around 115 days. With grain only diet periods of 120-130 days, they showed beef concentrations to be reduced by about 50%. The models of this assessment allow for the incorporation of an empirical reduction factor to account for a fattening program prior to slaughter. In the demonstration scenarios, it was assumed that the beef cattle slaughtered by the farmer for his home use were not fattened, and a value of 1.00 was assumed. For the air-to-beef model validation exercise, however, a value of 0.50 was applied, as suggested by these two research efforts.

The air-to-soil algorithms of the stack emission source category, and the soil-to-air algorithms of the soil contamination source categories have both been highlighted as algorithms which may have uncertainties. These uncertainties are detailed in Section 8.4.7. Generally, it was found that the air-to-soil algorithms may be slightly underestimating soil concentrations, while the soil-to-air algorithms may be underestimating air concentrations by an order of magnitude (although this speculation may not even be warranted, given that appropriate

experiments were not available to test the soil-to-air models). As a result, an examination of model trends show a key dichotomy in the way the stack emission source category performed as compared to the soil contamination source categories. Specifically, soil alone accounted for about 90% of the milk and beef impacts for the soil source category, whereas soil accounted for only about 5% of the milk and beef impacts for the stack emission source category. Refinements to the model algorithms or the model parameters which would increase air concentrations resulting from soils, and increase soil concentrations resulting from depositions would narrow this gap.

Data on rates of consumption of these food products, as well as the contact fractions used in the demonstration scenarios, were from the household component of the National Food Consumption Survey conducted by the USDA (USDA, 1992). A review of the uncertainties inherent in the use of this data is included in Section 8.4.7 above on fruit and vegetable ingestion, and will not be repeated here. One additional factor considered for meats of the terrestrial food pathways is the pre- and post-cooking losses, including factors such as weight loss by cooking, weight of bones, and so on. Based on data on such losses, consumption are reduced by about one-half based on these considerations.

A summary of uncertainties associated with the terrestrial animal food pathways is given in Table 8-8.

8.5. USE OF PROBABILISTIC TECHNIQUES FOR ASSESSING EXPOSURE TO DIOXIN-LIKE COMPOUNDS

The purpose of this discussion is to 1) briefly discuss how probabilistic techniques, such as Monte Carlo or Latin Hypercube simulations, work and could be applied in exposure assessments and 2) summarize recent efforts by five investigators to apply probabilistic procedures to assessments involving dioxin-like compounds.

Basically, Monte Carlo and Latin Hypercube assessments are generic statistical methods which generate a distribution for an output of a mathematical model using the distributions of the input variables. Computer simulations are used to repeatedly generate outputs based on parameter inputs, where values for parameters are selected from their distributions. The outputs are compiled and expressed as a frequency distribution. In the context of exposure assessment, for example, a Monte Carlo application could involve developing distributions for each of the parameters in the exposure equation and generating a distribution showing how the exposure levels vary in the exposed population. The final distribution can be interpreted as the probabilities of one individual (randomly selected from the exposed population) experiencing

various exposures. Since exposure levels are not only a function of the exposure parameters but also of the concentration in exposure media, another application of the Monte Carlo method would be to estimate the distribution of exposure media concentrations using mathematical models for fate and transport.

Probabilistic techniques can be a powerful tool for expressing variability and evaluating scenarios in exposure assessments. However, their use requires detailed input data which are frequently unavailable. Although the procedure may make an analysis look more elegant, it may actually yield misleading results if based on poor data. Accordingly, exposure assessors should be very cautious when trying to apply Monte Carlo techniques or interpreting the results.

Generally, Monte Carlo procedures should be applied only when credible distribution data are available for most of the key variables. Distribution data refers to empirical information on the statistical variation of the variable that is relevant to the site assessed. Usually this data should be obtained from surveys conducted at the site of interest. However, data on human behavioral characteristics could be obtained from survey information based on populations distant from the site, if comparability can be established.

Paustenbach et. al. (1992b) used Monte Carlo procedures to develop soil cleanup levels for 2,3,7,8-TCDD at residential and industrial sites. The following exposure pathways were included: dermal contact, soil ingestion, dust inhalation and fish ingestion. For each parameter a range of values was identified (on the basis of reported values in the literature) and a uniform distribution assumed. These assumptions are summarized in Table 8-9. For the residential scenario, the soil level corresponding to the 50th percentile (defined as 50% of the population being exposed below a risk of 10^{-5}) was 17 ppb and the 95th percentile was 7 ppb. For the industrial scenario (outdoors), the soil level corresponding to the 50th percentile was 160 ppb and the 95th percentile was 50 ppb.

Anderson et. al. (1992) used Monte Carlo procedures to describe the distribution of exposures to 2,3,7,8-TCDD occurring in various U.S. population segments as a result of ingesting fish caught near pulp and paper mills. The populations considered were all U.S. residents, all sport fishermen, U.S. residents living near (within 50 km) mills, and sport fishermen living near mills. The distributions for the various parameters were derived by either fitting idealized curves to empirical data or using personal judgement. These distributions are summarized in Table 8-10. The distribution of 2,3,7,8-TCDD concentrations in fish was derived from data collected in EPA's National Study of Chemical as exposure parameters. Distributions were developed for input factors and Monte Carlo Residues in Fish (EPA, 1992a). The following 50th and 95th percentile risks were estimated (using EPA cancer potency values):

all US residents - 1×10^{-9} & 3×10^{-7}
near mill residents - 4×10^{-8} & 2×10^{-6}
all sportfishermen - 2×10^{-8} & 3×10^{-6}
near mill sportfishermen - 6×10^{-7} & 2×10^{-5}

McKone and Ryan (1989) developed an exposure assessment procedure based on simple steady state transfer factors called PEFs or pathway exposure factors. These factors were applied to two paths: air/plant/food and soil/plant/food. This is an example of Monte Carlo techniques being applied to estimate exposure media concentrations as well as describe the variability in the distribution of exposure behaviors such as ingestion rates. The procedure was demonstrated using 2,3,7,8-TCDD and four pathways: ingestion of fruit/vegetables, grains, meat and dairy products. The distributions used for the various input parameters are summarized in Table 8-11.

Two recent assessments (Cullen, 1995; Keenen, et al., 1995) looked at the modeling of the impacts of dioxin-like compounds on indirect pathways from combustor emissions. Cullen (1995) looked the exposure to 2,3,7,8-TCDD through consumption of produce grown near a municipal solid waste incinerator. She included several parameters used to model the concentration of 2,3,7,8-TCDD in vegetables, and through a decomposition analysis, attempted to evaluate which parameters are most uncertain with regard to the modeling of vegetation concentration. Table 8-12 looks at the various parameters which Cullen (1995) evaluated to model vegetation concentrations.

Keenen, et al. (1995) evaluated another indirect pathway for 2,3,7,8-TCDD, also from an incinerator. The pathway they evaluated was the beef ingestion pathway, and the modeled incinerator type was a hazardous waste incinerator. They conducted a two-dimensional Monte Carlo exercise which separately characterized uncertainty and variation using a nested loop approach. They characterized the modeling of beef concentrations, including the modeling of the transfer of 2,3,7,8-TCDD from air to plant to animal, as uncertain, and characterized various exposure related quantities as variable. The “variable” considerations included locations of farms in relation to the incinerator (which was important because the air dispersion model predicted different concentrations and depositions of dioxins at sites of exposure), the individuals’ body weights, their beef consumption rates, and their exposure durations. In the nested approach, a beef concentration was estimated using the food chain model and parameter selections from probability density functions of the uncertain parameters. Then, a second nested procedure modeled a distribution of dose rates associated with the uncertainty calculation of the beef concentration. They results of their analysis suggest that exposures to 2,3,7,8-TCDD via

consumption of beef produced near a hazardous waste incinerator could have a total uncertainty spanning three orders of magnitude, and that the uncertainty was dominated by interindividual variability.

The five articles discussed above differ widely in how they have applied Monte Carlo methods, particularly in the selection of input parameter distributions. In some cases, it appears that uniform distributions were assumed due to the lack of data needed to support more complex distributions. The central values in these ranges probably occur more often than those near the ends, so the uniform distribution assumption probably underestimates the occurrence of central values and overestimates the occurrence of values near the ends of the distribution. Clearly more data are needed to better support input parameter distributions.

Also, the benefit of conducting Monte Carlo or other numerical methods to evaluate the uncertainty of model predictions of exposure media concentrations that result from a source of contamination is unclear. If attempting such an exercise, assessors must be aware of the following: 1) the relationship between contaminant fate parameters which are included in the same modeling exercise - high log Kow is associated with lower bioconcentration, lower volatility, etc.; and 2) the certainty in the range of parameters reported upon which a distribution is to be based - old literature, different and/or inappropriate experimental conditions, and so on. The authors of this assessment are of the opinion that the following exercises are far preferable in understanding and using fate and transport models for dioxins: 1) gaining confidence in a single set of dioxin fate parameters through model validation exercises, 2) checking the "validity" of predicted exposure media concentrations by comparing them with existing known concentrations, such as background concentrations or concentrations found in known settings of contamination, as a regular part of any fate modeling exercise, 3) understanding what optional models are available and, when possible, seeing if they result in a substantially different exposure media prediction, 4) identifying parameters of most uncertainty, and then determining how final predicted exposure media concentrations could vary as a result of varying that single parameters (as in the sensitivity analyses exercises in Chapter 6), and 5) compiling field data to assign and check on the all-important biotransfer/bioaccumulation parameters for the models of this assessment. These are exactly the exercises that have been undertaken in support of all the fate and transport models promoted in this document.

The five articles are just a small set of the growing body of literature on the topic of applying Monte Carlo methods to exposure and risk assessments. For example, the application of Monte Carlo methods to problems involving contaminated groundwater and related exposure pathways such as ingestion, indoor air inhalation and dermal contact with water has been

examined (McKone and Bogen, 1991). Although this work does not deal specifically with dioxin, it may be informative to readers generally interested in Monte Carlo procedures. Similarly, Paustenbach has published additional articles dealing with the application of Monte Carlo methods to exposure problems involving other chemicals (Paustenbach et al. 1991; Paustenbach, et al., 1992). Burmaster has also published numerous articles on this topic which may be of general interest to readers (ie. Burmaster and Stackelberg, 1991).

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Table 8-1. Uncertainties associated with the lifetime, body weight, and exposure duration parameters.

Assumption/ Method	Approach	Rationale	Uncertainty	Comments
Lifetime	70 yrs	Standard EPA assumption.	Actuary data indicate that lifetime may be increasing	Not a major source of uncertainty
Body weight	70 kg	Standard EPA assumption. Not needed for food pathways of fruit/vegetables, beef, milk, chicken and eggs because consumption rates are units of g/kg/day and hence incorporate body weight	Not much uncertainty. Current data suggests average body weights are lower and are different for men or women - averages above 60 kg for both.	Not a major source of uncertainty.
Exposure duration	9 and 30 years	Assumptions for central and high end exposure scenarios. Estimates are 50th and 90th percentile mobility survey results; higher estimate also justified based on the assumption that rural farming families live in one location longer than non-farming families in rural settings.	Can vary for site-specific applications. Source strength dissipation not a consideration for effluent discharge or stack emission sources assuming discharges/emissions continue for duration of exposure. However, source strength dissipation may be a consideration for soil contamination source.	The 30 year duration for high end farming families assumes such families are less transient than non-farming families.
Overall: Of these three parameters, the exposure duration is the most uncertain. The values used in this assessment were from mobility studies and they also considered that farming families may tend to live in one location longer than non-farming families. Evidence in the literature and a sensitivity analysis exercise in Chapter 6 suggest that soil concentrations of dioxins dissipate slowly, such that the assumption of non-reduction of the soil concentration over the duration of exposure for the soil contamination source category is a reasonable assumption.				

Table 8-2. Uncertainties associated with the soil ingestion pathway.

Assumption/ Method	Approach	Rationale	Uncertainty	Comments
Exposure site soil concentrations estimated when source is not at exposure site	Air dispersion and deposition modeled for stack emission source category; erosion modeled for soil contamination source category; for both, soil mixing depths of 2 and 20 cm, and 25 year half-lives assumed	Algorithms assume steady state; limited research suggests that the selected parameters are reasonable estimates for dioxin-like compounds.	A model validation exercise suggests that the deposition model is underestimating soil concentrations, most likely due to lack of consideration of vapor deposition and detritus additions. On the other hand, the soil erosion algorithm may be overestimating off-site impacts.	Future refinements should focus on improvements to air-to-soil modeling which add vapor impacts and detritus production.
Soil ingestion exposure assumptions	Ingestion occurs between 2-6 years old; central and high end rates of 100 and 600 mg/day; adult ingestion not considered	Soil ingestion is most likely to occur for this age range; ingestion rates selected from a review of studies in EPA (1997).	Adult soil ingestion could be important for farming situations where soil contact is frequent. Proper soil ingestion rates is an ongoing issue of research.	Ingestion rates do not consider pica behavior, which could lead to ingestion rates significantly higher than selected here. Adult soil ingestion should be considered for site specific application.
Overall: The modeling algorithms which are used to predict soil concentrations at a site of exposure when the source is distant from the site of exposure can result, in one case, an overprediction of soil concentrations, and in the other case, an underprediction of soil concentrations. The air-to-soil algorithm of the stack emission may be underestimating soil concentrations because of a lack of consideration of vapor phase impacts and a lack of consideration of detritus production. On the other hand, the erosion algorithm may be overestimating soil concentrations, based on a comparison of off-site impacts noted at an industrial site in Midland, MI, with the modeled off-site impacts. Uncertain parameters identified for soil concentration modeling include the soil dissipation rate (half-life of 25 years), the soil erosion and transport algorithm, the mixing depths, and for the stack emission source category, the uncertainties associated with the ISCST3 model. Soil ingestion for older children and adults was not considered. Assessors may wish to consider these pathways if soil concentrations at a site (modeled or measured) are high. Otherwise, soil exposure parameters are expected to be reasonable for general assessment purposes.				

Table 8-3. Uncertainties associated with the dermal exposure pathway.

Assumption/ Method	Approach	Rationale	Uncertainty	Comments
Soil concentration modeling	See Section 8.4.2 and Table 8-2 for a summary of the uncertainties in soil concentration modeling.			
Use of tillled vs. untilled concentrations	used "tilled" concentration for all outdoor dermal exposures and "untilled" for indoor dermal exposures	Soil dermal contact assumed to occur while farming/gardening outdoors; indoor dust assumed to originate from untilled soils.	Tilled concentrations are lower than untilled concentrations; no data available to show relationship between outdoor and indoor dioxin concentrations in soil/dust.	Assessors should consider site-specific behaviors to determine patterns of behavior leading to soil dermal contact
Contact/ adherence rate	0.005 mg/cm^2-event for indoor contact; 0.03 and 0.1 mg/cm^2-event for residential gardening and farming, respectively.	Based on data suggesting a much larger range from <0.002 to >20 (for "kids-in-mud"); measurement data available described in EPA, 1997	measurement data may have uncertainties; variability expected due to behaviors, soil type, and so on.	Should be considered an uncertain parameter, but little data is available to make better parameter assignments.
Contact frequency	365 events/yr for indoor; 350 events/yr for farming; 100 events/yr for gardening.	based on judgement and assumption that farmers spend more time in soil than non-farmers; indoor events judged to occur daily.	uncertainty judged to be relatively small given assumptions for behaviors	climatic conditions, behaviors, other site-specific factors could be important
Surface area	1,000 cm^2 for indoor events; 10,000 for residential gardening (central scenario) and 3600 for farming (high end scenario)	Based on total body surface area data and clothing assumptions: bare hands indoors; hands and arms for farmers; hands, arms, and legs for gardeners	Good data and small variability on body surface area; clothing assumptions based on judgement and site-specific conditiosn.	Studies have shown that fine particulates can deposit under clothing. Different behaviors can lead to different assumptions regarding exposed body areas.
Absorption fraction	0.03 for all compounds	Data in EPA (1992c) suggested a range of 0.001 to 0.03 based on data for 2,3,7,8-TCDD.	Value chosen was upper end of range, so refinements would appear to lead to lower estimates of dermal exposure.	Soil properties may also affect absorption.
Overall: The uncertainties and variabilities in the soil contact/adherence and absorption fraction parameters make the overall exposure estimates highly uncertain; judged to be plus or minus 1 to 2 orders of magnitude. Assessors should also be aware of uncertainties associated with prediction of soil concentration if source is distant from site of exposure.				

Table 8-4. Uncertainties associated with the water ingestion pathway.

Assumption/ Method	Approach	Rationale	Uncertainty	Comments
Water Concentrations	See modeling approaches for the soil, stack emission, and effluent discharge source categories in Chapter 4		literature data show few occurrences of dioxin-like compounds at 1 pg/L detection; models estimate 0.01 pg/L range and lower; cannot therefore ascertain uncertainty due to modeling, although little uncertainty expected due to low concentrations both found and predicted; and suitability of model predictions of sediment concentrations.	No major uncertainty expected due to modeling of water concentrations
Water Ingestion Rate	1.4 L/day central; 2.0 L/day high end	The classically assumed 2.0 L/day was evaluated as upper end rather than central value; 1.4 L/day recommended instead for central value.	EPA (1997) also noted that information on sensitive subpopulations such as laborers was unavailable; still, their analysis indicated that 2 L/day corresponds to high end value; hence it is appropriate for high end settings	Not expected to be a critical factor for uncertainty.
Contact Rate	0.70 central; 0.90 high end	values correspond to central and high end values for time at residence from several time use studies reviewed in EPA (1997).	The major uncertainty has to do with the extent to which exposed individuals rely on impacted water body for drinking water consumption. By using contact rates based on time at home, the assumption is that 100% of drinking water at home comes from impacted water body (which is assumed to supply water to household).	Exposure could be less if exposed individuals rely on water supply for drinking water other than impacted water body.
Overall: Data in the literature suggests concentrations mostly below 1 pg/L, which is consistent with modeling of concentrations 0.01 pg/L and lower in demonstration of all source categories. With this evidence, uncertainty with modeling is unknown, but the uncertainty is expected to be low because there is evidence that sediment predictions are consistent with field information. In general, water exposure is essentially the lowest exposure pathway.				

Table 8-5. Uncertainties associated with the fish ingestion pathway.

Assumption/ Method	Approach	Rationale	Uncertainty	Comments
Bioaccumulation approaches for fish tissue concentration estimation	Modeled bottom and suspended concentrations multiplied by BSAF or BSSAF	Bioaccumulation approaches rather than bioconcentration approaches are appropriate for lipophilic persistent organic compounds; water-based rather than sediment based approaches could be used, but sediment based approaches offer two advantages: 1) sediment data can and has been measured to derive field-based BSAF/BSSAFs - water concentrations for dioxins are too low, and 2) models for predicting sediment concentrations can likewise be tested.	Strictly speaking, BSAFs developed from one set of data are not transportable to other water bodies, and/or to other fish species; uncertainty exists with sediment concentration modeling; range of measured BSAF from selected value of 0.09 to greater than 1.00.	Model validations and comparisons of predicted with measured fish concentrations speak well for fate and transport algorithms
Fish ingestion rate	8 and 25 g/day for central and high end	Based on a review of recreational angler surveys in EPA (1997)	Assignment of this parameter should be based on site-specific considerations; subsistence behaviors leading to much higher fish ingestion rates should be ascertained.	Example settings were defined as rural/agricultural, but with a major river used as drinking water supply and suitable for fishing; hence, freshwater recreational fishing data was used.
Overall: Comparison of fish concentrations generated in the demonstration scenarios with literature values of fish concentrations of dioxin-like compounds shows them to be comparable. The validation using 104-mill data and testing the effluent discharge algorithms showed that fish concentrations were low by about one-half, but two important considerations for that test include: the discharging mill was assumed to be the only source of 2,3,7,8-TCDD, and uncertainties with the field data and the BSSAF lead to a conclusion that the model behaves quite adequately. Fish concentrations of PCBs may have been underestimated, but this conclusion is tempered by the fact that the modeled PCB sediment concentrations are similarly lower than has been measured. Alternate modeling approaches based on water column factors show comparable fish concentrations than sediment-based methods. Assignment of the fish ingestion rates was based on data from “recreational angler surveys”; “general population” and “subsistence” ingestion rates would be lower and higher, respectively, than selected for this assessment.				

Table 8-6. Uncertainties and sensitivities associated with estimating vapor and particle-phase air concentrations from contaminated soils.

Assumption/ Method	Approach	Rationale	Uncertainty	Comments
Exposure parameters	13 and 20 m^3/day for central and high end; 0.70 and 0.90 contact fractions for central and high end.	Central and high end estimates as described in EPA (1997)	not much uncertainty or variability expected for inhalation rates and contact fractions	uncertainty introduced by exposure durations of 9 and 30 years because of their role in the volatilization algorithm; otherwise uncertainty more an issue for methodologies estimating air concentrations.
Volatilization followed by near or far field dispersion for vapor phase concentrations	Used model developed by Hwang, et al (1986) for volatilization of PCBs; standard area source modeling for dispersion	Like PCBs, dioxin-like compounds are highly sorbed and persistent. Hwang (1986) model also has the advantage that the solutions were simplified using assumptions deemed reasonable for soils contaminated with dioxins.	Chemical parameters H and Koc are most uncertain with an order of magnitude range in estimated concentrations; estimations also sensitive to area, distance, and frequency wind blows to receptor.	An analysis of model performance suggests that the soil to air algorithms may be underestimating air concentrations, although this cannot be ascertained for certain because measured air concentrations are due to distant sources as well as soils.
Wind erosion followed by same dispersion algorithms	Used model based on highly erodible soils for dust flux to estimate fluxes for particle sizes < 10 µm	Assuming highly erodible soils may tend to overestimate flux, but not considering particles of size >10 µm would underestimate total airborne reservoir.	Parameters associated with the erodibility of soils can lead to a 2 order of magnitude range for estimated concentrations; much less sensitivity noted for other parameters.	No data to evaluate model results; however, particle concentrations are over 1 order of magnitude lower than vapor concentrations. Considering that the model was based on erodible soils, dust flux concentrations may be generally unimportant.

Table 8-6. (cont'd).

Assumption/ Method	Approach	Rationale	Uncertainty	Comments
Volatilization or resuspension of eroded contaminants not considered	Contaminants eroding to exposure site assumed not to volatilize or resuspend to contribute to exposure site concentrations	Traditionally, evaluation of the off-site impacts from a site of soil contamination considered only the impacts from the contaminated site.	If delivered contaminants volatilize or resuspend at site of exposure, exposure site air concentrations would increase by a factor of 2 to over 10.	More consideration of the fate of delivered contaminants is warranted.
Overall: The results of a model validation exercise showed that model predictions of air concentrations of dioxins resulting only from soil emissions were less than observed air concentrations by 2-3 orders of magnitude. The fact that they are lower is to be expected, since observed air concentrations over soils in an actual setting are very likely to be due mostly to long range air-borne transport from distant sources. Still, an analysis of data of plants growing in soils of known concentrations also suggests that the air-to-soil model may be underpredicting air concentrations. Ultimately, no data could be found on air concentrations over soils where it is definitely known that the soil is the only source of the dioxin-like compounds, so a degree of underprediction, if that is in fact occurring, could not be ascertained. Sensitivity analysis showed estimations of vapor phase air concentrations to be sensitive to Koc and H, and also to key source strength and delivery terms such as areas of contamination and wind speed. Assuming these non-chemical specific parameters can be known with reasonable certainty for site-specific applications, the most uncertainty lies with chemical specific data. Alternate approaches for volatilization and air dispersion generally estimate comparable air concentrations (within an order of magnitude or lower). Approaches to estimate particulate phase concentrations are empirical and based on field data. They are based on highly erodible soils but are specific to inhalable size particles, those less than 10 μm. As such, they may overestimate inhalation exposures, but may underestimate the total reservoir of particulates, which becomes critical for the particle deposition to vegetation algorithms. Another area of uncertainty is the assumption that volatilized contaminants do not become sorbed to airborne particles - this is also critical because vapor phase transfers dominate plant concentration estimation. A final key area of uncertainty is that transported contaminants from a contaminated to an exposure site via erosion are assumed not to volatilize or resuspend at the exposure site - air borne exposure site concentrations may be underestimated as a result.				

Table 8-7. Uncertainties associated with vegetable/ fruit ingestion exposure algorithms.

Assumption/ Method	Approach	Rationale	Uncertainty	Comments
Ingestion Rates and Contact Fractions	1.16, 1.52, and 1.47 g/kg/day root/above veg, and fruit; 0.101 and 0.173 for fruit and veg contact fractions	Derived from National Food Consumption Survey (NFCS) of 1987-88; Household portion. Available data allows for most appropriate estimates of homegrown rates and contact fractions, and also ties body weight to ingestion rate by expression in terms of g/kg/day.	Household survey relies on head of household recall for week long food brought into home; whole food product do not include cooking losses, discarded amounts, etc.	All parameters evaluated as reasonable for general exposure to categories of fruit and vegetables; more refinement desired for specific assessments.
Below ground vegetable concentration	Used empirical RCF, root concentration factor, based on Kow, and VG_{bg}, below ground correction factor	Approach based on laboratory experiments; validation exercise on data for carrot peel concentrations grown in soils of known concentrations supports model capabilities.	VG_{bg} of 0.25 based on evidence of some translocation into carrots and potatoes; however, it remains most uncertain parameter; Kow is also uncertain, although validation exercise supports use of RCF	Further refinements to VG_{bg} may be warranted.
Above ground vegetable, fruit concentration	Air-to-leaf vapor phase transfer algorithm based on B_{vpa} (transfer factor) which was developed from field data; vapor phase impacts also include VG_{ag}; particle deposition algorithm for particle bound dioxins	Field experiments and modeling both show that vapor phase impacts dominate total plant concentrations; B_{vpa} calibrated from field data; particle deposition algorithm developed for radionuclide impacts to agriculture; VG_{ag} assignment of 0.01 considers both evidence of little within plant translocation for exposed bulky vegetations and reduction in plant concentrations due to peeling prior to consumption.	Model validation/comparison exercise showed the air-to-leaf model to work reasonably well in rural setting, but to underestimate grass concentrations when grass was grown in contaminated soils in an industrial setting; VG_{ag} still the most uncertain parameter.	Limited literature data and model validation exercise suggests that above ground vegetative impacts from contaminated soils may be underestimated; could be due to lack of consideration of rainsplash.
Overall: All ingestion parameters assumed are evaluated as reasonable for general exposure to broad categories of fruits and vegetables. However, great variability is expected if using these procedures on a specific site where home gardening practices can be more precisely ascertained. Validation exercises support both the soil to below ground vegetable and air-to-leaf algorithms. The most uncertain parameters for both algorithms are the "VG" parameters, VG_{ag} and VG_{bg}, which correct for evidence that there is little within plant translocation of dioxins in below as well as above ground bulky vegetations, and additionally considers peeling or washing of vegetations, which would further reduce whole plant concentrations.				

Table 8-8. Uncertainties associated with the terrestrial animal food pathways.

Assumption/ Method	Approach	Rationale	Uncertainty	Comments
Ingestion rates, contact fractions, food preparation consideration	Like for fruit/vegetables, ingestion rates and contact fractions developed from the household component of the National Food Consumption Survey (USDA, 1992; as interpreted in EPA, 1997). For meats, an additional pre- and post-cooking factor of about 0.50 further reduces consumption rates derived from this data to better relate food "brought into the house" to food "as eaten".			
Terrestrial animal bioconcentration models	Weighted average concentration in diet times BCF equals fat concentration; BCF for beef/milk developed from one experiment on one lactating cow.	Precedence and data support this approach. Two field studies collecting data to develop BCF for cow's milk arrive at very similar BCFs.	Uncertainty in applying milk-derived BCF to beef; chicken and egg BCFs separately derived in laboratory experiments; uncertainty in applying laboratory feeding experiments to field situations for chickens; dietary assumptions are variable and soil bioavailability correction factor, Bs, is uncertain and important for free range chicken scenario	Air-to-beef model validation exercise supports the use of milk-derived BCFs for beef, and approach in general. For predicting beef concentrations, site-specific consideration of fattening regime is important.
Related models	See previous sections for discussions on uncertainties in associated models including air dispersion and deposition modeling, soil-to-air and air-to-soil modeling, air-to-plant modeling, and soil concentration modeling.			
Overall: The demonstration scenarios showed that the terrestrial animal food pathways dominate human exposure. This was supported by similar findings in Volume III of this assessment, which estimated background exposures based on measured concentrations coupled with consumption rates. In site-specific applications, animal diet fractions in the various categories of animal feeds (leafy, partially protected, fully protected, soil) becomes important. The air-to-beef model validation exercise described earlier lends confidence to the use of the milk/beef bioconcentration algorithms.				

Table 8-9. Distributions for a Monte Carlo exercise which developed soil cleanup levels at residential and industrial sites.

Parameter	Range (Residential)	Range (Industrial)
Soil Contact µg/cm^2/d	200 - 1800	same
Dermal Bioavailability Fraction	0.01 - 0.025	same
Fraction soil from site	O-5 yr: 0.1 - 1.0 6-30 yr: 0.1 - 0.5	0.1 - 1.0
Fraction indoor dust contaminated	(not considered)	0.25 - 1.0
Indoor exposure duration	0-1.5 yr: 182-365 d/yr 1.5-30 yr: 200-365 d/yr	0 - 8 hr/d 220 - 260 d/yr
Outdoor exposure duration	0 - 1.5 yr 60-120 d/yr 1.5 - 30 yr 60-240 d/yr	0 - 8 hr/d 220 - 260 d/yr
Soil ingestion rate, µg/d	0 - 1.5 yr 100 - 10000 1.5 - 5 yr 9000 - 50000 6 - 12 yr 5000 - 50000 13 - 30 yr 100 - 50000	100 - 50000 (indoors) 100 - 10000 (outdoors)
Oral Bioavailability	0.38, 0.40, 0.47, 0.49	same
Air particulate concen., µg/m^3	25 - 45	same
Fraction outdoor dust contaminated	0.1 - 0.5	same
Inhalation rate m^3/hr	0-1.5 yr: 0.03 - 0.07 1.5-5 yr: 0.3 - 0.9 6-12 yr: 0.75 - 1.5 13-30 yr: 0.5 - 1.5	9 - 14.6 m^3/d
Lipid Content of Fish	0.01 - 0.05	
Fish Bioavailability Index	0.01 - 0.5	
Organic Carbon content of sediment	0.01 - 0.5	
Fish Consumption, g/d	0-1.5 yr: 0 1.5-5 yr: 0.38 - 0.62 6-12 yr: 0.63 - 1.0 13-30 yr: 1.1 - 1.8	
Fraction remaining after cooking	0.3 - 0.75	

Source: Paustenbach et. al. (1992a); uniform distributions assumed over ranges shown.

Table 8-10. Summary of Monte Carlo distributions used in a fish consumption assessment.

Exposure Parameter	Distribution Type	Mean	Standard Deviation	Min./Max.
Dioxin Conc. (ppt of TEQ)	truncated lognormal	3.3	8.7	0.0002 /16,000
Fraction caught in affected waters	triangular	0.09 (all US) 0.4 (near mill)	0.2 0.2	0/1.0 0/1.0
Consumption (g/d)	truncated lognormal	2.5 (all US) 19.1 (sport - fishermen)	7.3 27.9	0/240 0.2/403
Duration (yr)	truncated lognormal	13.3	12.3	0.1/70
Cooking Loss Fraction	uniform	0.1	0.3	0.25/0.75
Body Weight (kg)	normal	71	18.1	29.9/143.2

Source: Anderson et. al. (1992).

Table 8-11. Summary of Monte Carlo distributions used in food chain study.

Parameter	Geometric Mean	Geometric Standard Deviation	Distribution
Milk Ingestion 0-15 yr: kg/kg/d 15-70 yr:	0.014 0.0033	1.2 1.1	log normal
Meat Ingestion 0-15 yr: kg/kg/d 15-70 yr:	0.0044 0.0029	1.1 1.2	log normal
Fruit/Veg Ing. 0-15 yr: kg/kg/d 15-70 yr:	0.0081 0.0045	1.4 1.3	log normal
Grain Ing. 0-15 yr: kg/kg/d 15-70 yr:	0.0074 0.0030	1.2 1.2	log normal
Particle to Food Deposition Factor, m/d	300	3	log normal
Plant/Soil Part. Factor	0.013	4.0	log normal
Biotransfer Fac. Cattle Intake to Meat, d/kg	0.055	3.0	log normal
Biotransfer Fac. Cattle Intake to Milk, d/kg	0.01	3.0	log normal
	Lower Bound	Upper Bound	
Annual Inventory Food Crops, kg/m^2	1.0	10.0	log uniform
Annual Inventory Pasture Crops, kg/m^2	0.1	1.0	log uniform
Weathering Rate Constant, 1/d	0.01	0.1	log uniform
Cattle Inhalation Rate, m^3/d	63	177	uniform
Beef Cattle Ingestion of Pasture Grass, kg/d	4.0	20	uniform
Dairy Cattle Ingestion of Pasture Grass, kg/d	11	23	uniform
Cattle Soil Ingestion, kg/d	0.1	0.83	uniform

Source: McKone and Ryan, 1989.

Table 8-12. Summary of parameter distributions used for modeling terrestrial fruits and vegetables for human consumption in a Monte Carlo exercise.

Definition	Forms and Parameters	Distribution
I. Plant Concentration Modeling		
soil decay constant, 1/day	0.0001-0.0002	Uniform
soil mixing depth, m	0.15-0.25	Uniform
crop interception fraction	root, vine, tree: 0.05-0.25 Leafy: 0.16-0.40	Uniform
soil bulk density, g/m^3	median = 1.4; geo. stan. dev = 1.15	Lognormal
growing season duration, days	tree: 120-150 leafy: 40 - 60	Uniform
root uptake factor for translocation of TCDD from soil to crop, C_{plant}/ C_{soil}	root crop: 0.25-1.0 Vine, leafy: 0.05-0.15	Uniform
deposition velocity of particle or vapor class, m/day	median by diameter, variability = 0.8 log units	Loguniform
particle weather rate constant on plant, 1/day	0.01-0.1	Loguniform
crop yield, g/m^2	vine, tree: 5 - 15 leafy: 6 - 10	Uniform
II. Air Concentration Modeling		
dispersion factor calculated from air dispersion model, defined as air concentration in ith sector per unit mass emitted, $(mg/m^3)/(mg/s)$	median = -0.00017, geo. stan. dev = 1.4	Lognormal
2,3,7,8-TCDD mass emissions, mg/sec	$1.5*10^{-6}$ - $5*10^{-6}$	Loguniform

Source: Cullen (1995b)